Payments for Hydrological Ecosystem Services in Integrated Water Resources Management

Zahlungen für hydrologische Ökosystemleistungen im Kontext eines Integrierten Wasserressourcen Managements

Vom Fachbereich Bau- und Umweltingenieurwissenschaften zur Erlangung des akademischen Grades Doktor-Ingenieur (Dr.-Ing.)
genehmigte Dissertation von Dipl.-Ing. Jochen Hack aus Kaiserslautern
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Foreword

The awareness that an integrated well co-ordinated management of water resources is the ultimate sustainable approach has a long history. This has led to the global paradigm of Integrated Water Resources Management (IWRM). One of the connected problems is the choice of adequate temporal and spatial scales. Worldwide, concerned authorities propagate the water management on a river basin scale; very early concepts were realised in Central Europe which were harmonised in the European Water Framework Directives 60/2000. Increasingly these approaches are also introduced in developing countries.

However, without neglecting the positive progress achieved all concepts known lack generally applicable approaches for the integration of land, energy and water resources on regional and local scale - for a good reason: The enormous interdisciplinary and transdisciplinary complexity of the problem. This has found the interest of the scientific and public community involved and a very large number of recent publications.

In conclusion, a critical dilemma is the fact that the generally accepted global approach needs to be implemented on a local and regional scale determined by political i.e. community boundaries. Local and regional actors, however, have been rarely part of theoretical considerations.

Somehow independent from IWRM the concept of payments for hydrological ecosystem services (PHES) has developed aiming at the stabilisation of ecosystem functions to maintain or provide water for different purposes, ranging from environmental protection to concrete provision of water for downstream users. It is overdue to integrate the PHES concept into IWRM.

Jochen Hack has recognised the need for a common knowledge base. Without claiming to be complete he analysed the existing highly interdisciplinary and transdisciplinary literature with more than 600 contributions, filtered out similarities and contradictions and drew valid conclusions. Based on this scientific analysis he developed his own ideas and concepts for a combined view and practical approach. This work was finally enriched by a case study in Nicaragua which he observed and accompanied for several years.

The supervison of Jochen Hack's thesis is my last active contribution during my academic career, which gave me a great pleasure. I hope his convincing thesis will also be the basis for his own academic career. I also hope that he will continue to work on timely interdisciplinary teaching and research concepts which are so much harder to develop and implement than pure disciplinary ones.

Perdifumo, Italy, April 2014

Manfred Ostrowski

Abstract

This dissertation documents, analyzes and interprets the state of knowledge of the global Integrated Water Resource Management (IWRM) implementation process. Based on a comprehensive assessment of global implementation reports and the scientific discourse of two decades general problems of implementation and specific constraints to the process in developing countries are identified. Moreover, present IWRM trajectories and recommendations towards improvements of implementation and operationalization are derived from scientific and practitioner's experience. Hence, the principal contribution of this dissertation in this basic problem analysis is a detailed clarification of the problem statement of IWRM implementation in developing countries and the provision of guiding principles for improvements.

Identified solution approaches are further conceptualized methodologically by applying the theoretical concept of institutional fit and interplay. In the following, this dissertation asserts that problems of institutional fit and interplay need to be solved interdependently with due regard to specific operational constraints of local contexts in developing countries in order to successfully implement IWRM at an operational management level, i.e. operationalization of IWRM. Hence, this dissertation compares different policy instruments to achieve this based on an actor-centered incentive approach. It is argued that improvements require a mix of policies with specific instruments to enhance institutional fit and interplay suitable for context-specific operational constraints. This dissertation exposes that traditional command and control approaches for IWRM operationalization are insufficient and complementary instruments are necessary to address the prevailing implementation gaps. The instrument of Payments for Hydrological Ecosystem Services (PHES) is identified as a potentially suitable policy instrument because it combines several beneficial characteristics of communication / diffusion and economic instruments with characteristics of collaborative agreements. Applying the concept of institutional fit and interplay in the context of operational constraints of developing countries to identify suitable policy instruments for improved IWRM operationalization, as done in this dissertation, is a particularly novel approach.

In order to assess the potential contributions of PHES schemes to the operationalization of IWRM, the concept of ecosystem services and their valuation as its theoretical basis, are examined concerning the solutions it provides to address the governance challenges of institutional fit and interplay. In this context, hydrological ecosystems services and the their valuation are addressed specifically. Thus, this dissertation contributes to understanding how the application and valuation process of hydrological ecosystem services inherently define spatial relationships between potential service providers and beneficiaries based on functional ecosystem linkages. Subsequently, it is illustrated how the concept can positively influence the identification of appropriate context-specific scales for operational IWRM as a result of fit and interplay interdependence. Additionally, a cross-sectoral and cross-jurisdictional integration effect of the concept is acknowledged. The concept of hydrological ecosystem services has not yet been considered in relation to problems of fit and interplay in IWRM implementation as presented in this work. Hence, this dissertation provides additional insights in this regard.

In a further theoretical analysis, this dissertation documents, analyzes and interprets the state of knowledge of the economic conceptualization of the PHES instrument. It brings forward supporting arguments for a less market-based and therefore a stronger multi-faceted incentive-based interpretation of PHES. Based on a broad meta-analysis of global and regional PES scheme assessments, the principal characteristics of the instrument and its implementation are identified. A comprehensive instrument characterization of this kind is a further significant contribution of this dissertation which has not been done yet to a similar extent. The characterization and instrument assessment provides important insights with regard to the typical roles of different actors and the provision of incentives for behavioral change towards IWRM. Moreover, it contributes to understanding how the instrument can potentially address institutional challenges of IWRM operationalization in the context of general operational constraints. However, locally user-(co-)financed PHES schemes as a particular type were identified as especially conducive to engaging stakeholders and to promoting public participation.

Finally, the role of the PHES instrument in the context of a national IWRM process is assessed based on an empirical example taken from Nicaragua. This dissertation provides a comprehensive documentation of the national IWRM process in Nicaragua and its principal implementation gaps. The generalization of implementation gaps and specifically operational constraints made before can be confirmed for the Nicaraguan IWRM process. Moreover, the shortcomings of a formal top-down implementation approach based on command and control instruments alone are highlighted as well. Further valuable findings can be derived from this dissertation for other developing countries with a similar IWRM process through the analysis of contributions of locally user-(co-)financed PHES schemes to solve the problems of institutional fit and interplay in Nicaragua. Additionally, this dissertation reveals how the PHES instrument fits into an existing policy mix in Nicaragua and how it interacts with traditional regulations.

Hence, this work provides guidance on how to improve context-specific fit and horizontal interplay at the operational level of IWRM as well as on how to complement the primarily top-down directed IWRM implementation from bottom-up. Hence, this dissertation documents that the PHES instrument is more than a tool to finance nature conservation. Indeed, it shows that the implementation and execution process of PHES schemes fulfills several other tasks which are essential for the operationalization of IWRM.

Zusammenfassung

Die internationale Gemeinschaft hat sich das Integrierte Wasserressourcen Management (IWRM) zum Ziel gesetzt, um der übergeordneten Bedeutung der Wasserressourcen als integrierende Landschaftkomponente und für die sozio-ökonomische Entwicklung der Menschheit Rechnung zu tragen. Bisher konnten die Ziele dieser integriert und partizipativ organisierten Bewirtschaftung auf Flusseinzugsgebietsebene, insbesondere in Entwicklungs- und Schwellenländern, jedoch noch nicht zufriedenstellend erreicht werden. Eine Folge davon ist u.a. die fortschreitende Degradierung von Ökosystemen und damit ein Verlust der Dienstleistungen, die diese der Gesellschaft erbringen. Eine Vielzahl von Initiativen bemüht sich um die Inwertsetzung dieser, bisher als kostenlos wahrgenommenen, Ökosystemdienstleistungen und der Etablierung von Zahlungssystemen, um deren Erhalt zu finanzieren. Besonders Zahlungssysteme für hydrologische Ökosystemdienstleistungen werden als eine vielversprechende Alternative zu traditionellen Umweltpolitikinstrumenten angesehen, um externe Effekte nicht nachhaltiger Landnutzung zu adressieren. Der Ökosystemansatz verbindet sie dabei mit dem IWRM-Prozess.

Die im Rahmen dieses Dissertationsvorhabens realisierten Untersuchungen befassen sich mit dem Anwendungspotenzial von Zahlungssystemen für hydrologische Ökosystemdienstleistungen im Kontext eines Integrierten Wasserressourcen Managements in Entwicklungs- und Schwellenländern.

Aufbauend auf einer Analyse des Entwicklungs- und Implementierungsprozesses von IWRM im globalen Kontext (Kapitel 2) werden zunächst Kritik und Hürden in Bezug auf den Implementierungsprozess diskutiert. Weiterhin wird der aktuelle Stand der Forschung und Praxis in Hinblick auf die Fortentwicklung des IWRM-Konzepts und aktueller Implementierungsansätze erörtert.

Der aktuelle Stand der Forschung zum IWRM-Implementierungsprozess (Hürden und aktuelle Implementierungsansätze) bilden den Rahmen für eine strukturelle Konzeptualisierung der Implementierungsproblematik in Kapitel 3. Im Rahmen der Konzeptualisierung werden basierend auf der *Fit und Interplay* - Theorie generelle institutionelle Anforderungen und konkrete operationelle Rahmenbedingung in Bezug auf Entwicklungs- und Schwellenländer erarbeitet. Abschließend werden mögliche Kategorien von Steuerungsinstrumenten beschrieben und konkrete Anforderungen an IWRM-Implementierungsinstrumente abgeleitet.

Kapitel 4 beschreibt zunächst theoretisch das Konzept der Ökosystemleistungen im Allgemeinen als Lösungsansatz, während Kapitel 5 anschließend Zahlungssysteme für hydrologische Ökosystemleistungen im Speziellen als potenzielles Instrument zur IWRM-Implementierung behandelt. Im Rahmen einer Auswertung des prinzipiellen Implementierungsvorgangs international angewandter Zahlungssystemen für hydrologische Ökosystemleistungen wird die grundlegende Eignung des Instruments in Hinblick auf die in Kapitel 3 entwickelten Anforderungen an IWRM-Implementierungsinstrumente in einem ersten Analyseschritt beurteilt. Dabei wird insbesondere untersucht, ob die genannten Zahlungssysteme mit den Prinzipien und Zielen des IWRM übereinstimmen, sie eine zweckmäßige Ergänzung zu bestehenden umweltpolitischen Instrumenten darstellen und ferner den IWRM-Prozess auf Einzugsgebietsebene begünstigen bzw. fördern. Es ist somit die funktionale Rolle der Zahlungssysteme für hydrologische Ökosystemdienstleistungen Untersuchungsgegenstand sowie die Synergien bei der Implementierung solcher Zahlungssysteme mit dem lokalen IWRM-Prozess.

In einem weiteren Untersuchungsschritt (Kapitel 6) wird die Eignung des Instruments zur Unterstützung des IWRM-Implementierungsprozess am Beispiel Nicaraguas untersucht. Nicaragua hat in den letzten Jahren wichtige Schritte für einen nationalen IWRM-Prozess eingeleitet, die Umsetzung der IWRM-Prinzipen zum nachhaltigen Flusseinzugsgebietsmanagement auf lokaler Ebene steht jedoch

noch aus. Anhand eines Fallstudienvergleichs einzelner, voneinander unabhängiger Projekte von Zahlungssystemen für hydrologische Ökosystemleistungen, sowie einer intensiven Prozessbegleitung bei der Implementierung einer Fallstudie, wird die funktionale Rolle der Zahlungssysteme untersucht und die Bedeutung des Instruments für den nationale IWRM-Kontext erörtert.

Die Arbeit schließt mit einem Fazit und Ausblick zu weiterem Forschungsbedarf in Kapitel 7 ab. Es werden Chancen und Einschränkungen der Anwendung von Zahlungssystemen für hydrologische Ökosystemleistungen zur Implementierung von IWRM im Rahmen der Diskussion der Ergebnisse der Kapitel 5 und 6 analysiert und ausgewertet.

Danksagung

Die vorliegende Arbeit ist während meiner Tätigkeit als wissenschaftlicher Mitarbeiter am Fachgebiet Ingenieurhydrologie und Wasserbewirtschaftung des Instituts für Wasserbau und Wasserwirtschaft der TU Darmstadt entstanden.

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Ein ganz besonderer Dank gilt meiner Frau, meiner Familie und meinen Freunden, die mich zu jeder Zeit unterstützt haben. Diese Unterstützung hatte einen maßgeblichen Anteil am erfolgreichen Abschluss dieser Arbeit.

Darmstadt, im April 2014

Jochen Hack

Contents

List of Figures

List of Tables

List of abbreviations

CBD	Convention on Biological Diversity
CDM	Clean Development Mechanism
CICES	Common International Standard for Ecosystem Services
CM	Choice Modeling
CSD	Commission on Sustainable Development
CV	Contingent Valuation
EEA	European Environment Agency
ES	Ecosystem Services
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
GIZ	German Agency for International Cooperation
GWP	Global Water Partnership
HDI	Human Development Index
HP	Hedonic Pricing
ICLEI	International Council for Local Environment Initiatives
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
IRBM	Integrated River Basin Management
IUCN	International Union for Conservation of Nature
IWMI	International Water Management Institute
IWRM	Integrated Water Resources Management
JPI	Johannesburg Plan of Implementation
MBI	Market Based Instruments
MDG	Millennium Development Goals
MEA	Millennium Ecosystem Assessment
MES	Markets for Ecosystem Services
OECD	Organisation for Economic Co-operation and Development
NGO	Non-Governmental Organization
PASOLAC	Programa para la Agricultura Sostenible en Laderas de América Central
PES	Payments for Ecosystem Services
PHES	Payments for Hydrological Ecosystem Services
RBC	River Basin Council
RBO	River Basin Organization
REDD	Reduced Emission from Deforestation and Degradation
SADC	Southern African Development Community
SES	Social-Ecological System
TAC	Technical Advisory Committee of the Global Water Partnership
TCM	Travel Cost Method
TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total Economic Value
TNC	The Nature Conservancy
UN	United Nations
UNCED	United Nations Conference on Environment and Development
UN-DESA	United Nations Department of Economic and Social Affairs
UNDP	United Nations Development Programme
UNECE	United Nations Economic Commission for Europe
UNEP	United Nations Environmental Programme
UNSD	United Nations Statistical Division
WFD	European Union Water Framework Directive
WTA	Willingness To Accept
WTP	Willingness To Pay
WWAP	World Water Assessment Programme
WWC	World Water Council
WWDR	World Water Development Report

1 Introduction to the problem context and relevance of the investigation topic to be investigated

The subordinate importance of water for humans and nature as well as its complex interconnections with other media such as air and soil has led to the recognition that the management of water resources requires a holistic approach. In contrast to the sectoral approach to water management taken in the past, the international community now broadly agrees on the need for a general concept aiming at the integration of different environmental media across all sectors of society that use or have an impact on water resources. This concept, known as Integrated Water Resources Management (IWRM), has become a global paradigm for sustainable water resources management.

However, efforts to implement the concept started in the 1990s, but after two decades successful steps towards implementation remain limited mainly to developed countries, e.g. member states of the European Union (EU) implementing the European Union Water Framework Directive (WFD) or Integrated Catchment Management in Australia and New Zealand. Despite being recognized as a substantial means to achieve the Millennium Development Goals (MDG) and sustainable development in general, IWRM still needs to be successfully implemented in most developing countries. The global Status Report on the Application of Integrated Approaches to Water Resources Management from 2012, for instance, stresses that since 1992, when IWRM was postulated as part of the Agenda 21 as a paradigm for sustainable water resources management, 80 % of all countries of the world have embarked on reforms to improve the enabling environment for water resources management based on the application of integrated approaches (UNEP, 2012). Despite these considerable efforts in the establishment of enabling environments for IWRM through new water legislation, there has been much less success, especially in developing countries, in the management of water resources on the basis of the river basin, the cross-sectoral involvement of stakeholders, the establishment of participation mechanisms and integrated environmental conservation.

Implementation of IWRM has been hindered, on the one hand, because there is no consistent definition that can be made operational with measurable criteria (Van der Zaag, 2005; Jeffrey and Gearey, 2006; Biswas, 2008; Medema et al., 2008; Merrey, 2008; Molle, 2008b; Cardwell et al., 2009). Despite the controversy with regard to its proper definition, IWRM is in essence an approach to water management that seeks to integrate physical systems with human systems to shift away from fragmented planning. On the other hand the constraints of real-life political, social, and physical factors make IWRM difficult to achieve in practice without considering the specific natural and human system contexts (Pahl-Wostl et al., 2007; Merrey, 2008; Lankford et al., 2007; Medema et al., 2008; Moss and Newig, 2010; Butterworth et al., 2010; Beveridge and Monsees, 2012). This is partly due to the fact that IWRM implementation is an inherently complex task requiring additional financial and personal resources as well as new forms of governance. Adequate resources are often lacking in developing countries and natural resource management takes second place compared to economic development. Moreover, the IWRM paradigm is based on coordination, cooperation and integration. The implementation of IWRM requires supplementary actions in the way how water and land resources are managed. Although widely accepted and with several requirements for implementation already taken, the progress of IWRM on the river basin level is still not satisfactory. A recent study of the Organisation for Economic Co-operation and Development (OECD) on IWRM implementation, thus, comes to the conclusion “while the need for integrated water policy is widely acknowledged, it has not yet totally been achieved on the ground. In many countries, water governance is still in a state of confusion. In both developing and developed nations, water policy, to a greater or lesser degree, intrinsically raises governance challenges” (OECD, 2011b).

Traditional environmental governance has not led to the desired result in this respect, thus alternative governance elements seem necessary.

In the field of environmental conservation the ecosystem services concept has gained substantial assent within a broad community of scientist, politicians and different societal organizations. The Millennium Ecosystem Assessment (MEA) in 2005 strongly influenced the increasing recognition of ecosystems as potential providers of services essential for human well-being (MEA, 2005). The Economics of Ecosystems and Biodiversity (TEEB) study launched by Germany and the European Commission in 2007 takes a further step by drawing attention to the global economic benefits of biodiversity and ecosystems in order to enable practical policy actions. Further evidence of the relevance of the concept of ecosystem services and biodiversity is reflected in the recent foundation of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) in April of 2012 which, analogous to the Intergovernmental Panel on Climate Change (IPCC) for climate change issues, “will respond to requests for scientific information related to biodiversity and ecosystem services from governments” (IPBES, 2012).

Unsurprisingly, there have been several attempts to develop policy instruments that aim to account for the economic value of ecosystems and the services they provide. Payment or compensation for Ecosystem Services have evolved as a promising policy instrument to improve environmental conservation based on the valuation of ecosystem services (Mayrand and Paquin, 2004). Especially in the context of watershed protection, several attempts have been made on different policy levels to achieve this goal through Payments for Hydrological Ecosystem Services (PHES).

Although already implemented at several sites, PHES are still an ongoing subject of investigation. Current research on PHES is mainly focused on efficiency or equity aspects, to a much smaller extent on bio-physical relationships of land use changes and the provision of ecosystem services. Often the focus is on efficiency concerns rather than other environmental policy action, and the valuation of ecosystem services itself is also a matter of investigation. The consideration of PHES in the context of IWRM has gained much less attention and if so PHES are usually assessed as an alternative policy instrument, without thorough examination of the broader functional role that PHES may play in the context of IWRM establishment or improvement on the river basin level. Hence, from a water management perspective within IWRM little progress has been made on the assessment of PHES.

General aim of this thesis is to investigate how and to what extent PHES fits in the IWRM process and what role PHES can potentially play to further operationalize IWRM. Against this background, this dissertation aims to address the following aspects (structural role of the aspect within the context of the dissertation is stated in parenthesis) :

- Why is IWRM still not widely implemented, despite having been promoted strongly by the international community over the past three decades?
  (Problem analysis and general problem statement)
- What are the principal implementation gaps and core problems of the concept's operationalization in developing countries? What are the specific requirements for instruments to improve IWRM implementation?
  (Methodological conceptualization of the implementation problem)
- Payments for Hydrological Ecosystem Services (PHES) have gained significant popularity in the field of natural resources management in developing countries. Do the ecosystem services concept and PHES, in general, bear potential to contribute to improvements in IWRM implementation?
  (General theoretical analysis of a solution statement)
- What are the contributions of PHES schemes in a specific empirical context of IWRM implementation?
  (Specific empirical analysis of the solution statement)
- Can general recommendations be made on how PHES may contribute to the IWRM process?
  (Generalization of results)

1.1 Structure of the thesis and applied methodology

The general structure of this dissertation consists of five principal parts: a definition of problem statement and identification of the state of the art (Chapter 2), a methodological conceptualization and the development of a solution statement (Chapter 3), a general theoretical analysis of the proposed solution statement (Chapters 4 and 5), a specific empirical analysis of the solution statement (Chapter 6), and finally a presentation and assessment of the main results (Chapter 7). Figure 1.1 illustrates the general structure with its principal parts and corresponding chapters.

Problem statement and analysis

Chapter 2: Evolution and implementation of Integrated Water Resources Management: State of the art of implementation, scientific discourse on the concept, identification of implementation gaps and solution approaches.

Methodological conceptualization and solution statement

Chapter 3: Methodological conceptualization of IWRM implementation problems: Identification of the core problems of IWRM implementation, identification of requirements on implementation instruments.

General theoretical analysis

Chapter 4: The concept of Ecosystem Services and their valuation: State of the art of the Ecosystem Services approach and valuation methods, potential to improve IWRM implementation at the governance level.

Chapter 5: The concept of Payments for Hydrological Ecosystem Services: Analysis of Payments for Hydrological Ecosystem Services as a potential instrument to improve IWRM operationalization at the management level.

Specific empirical analysis

Chapter 6: Payments for Hydrological Ecosystem Services and IWRM in Nicaragua: Analysis of the contribution of the implementation process of Payments for Hydrological Ecosystem Services to the operationalization of IWRM in Nicaragua.

Presentation of results and assessment

Chapter 7: Conclusions

Figure 1.1: General structure of the dissertation

The assessment of the functional role of Payments for Hydrological Ecosystem Services (PHES) in the context of Integrated Water Resources Management (IWRM) is the general subject of this dissertation.

In a first step, the historical evolution and main principles of IWRM are analyzed. Based on this a definition of problem statement and identification of the state of the art concerning the implementation and operationalization of IWRM is accomplished. Thus, Chapter 2 summarizes the historical development of IWRM departing from traditional water resources management dominated by separated sector policies. It highlights the underlying motivation for IWRM and the present status of IWRM implementation while focusing on prevailing obstacles. At the end of Chapter 2 the most recent developments of the IWRM concept and possible improvements in its implementation are discussed in order to develop a general problem statement.

Based on the findings on the state of the art of the IWRM implementation process, a methodological conceptualization of the core implementation problems of IWRM is carried out in Chapter 3. The conceptualization of the problem of IWRM implementation considers the two general governance problems of achieving appropriate institutional fit and interplay. Furthermore, the chapter includes an analysis of general operationalization constraints to IWRM in developing countries at the management level. Following this, a comparison of contributions made to the operationalization of IWRM by different policy instruments is realized on the basis of the methodological conceptualization and the identified operationalization constraints. The chapter concludes with the identification of Payments for Hydrological Ecosystem Services as a potential instrument to improve IWRM operationalization in developing countries.

Chapter 4 takes up the ecosystem approach with a brief introduction of the ecosystem services concept especially in the context of environmental and biodiversity conservation policy, including the principle theoretical basis of ecosystem processes, functions and service provision. Furthermore, the valuation and prevailing state of the art valuation methods are reviewed. Following this documentation of the state of the art of the ecosystem services approach and methods of valuation with special focus on hydrological ecosystem services, an analysis of potential contributions of the approach to address the core governance problems of fit and interplay in the context of IWRM implementation is carried out.

The following chapter 5 deals with Payments for Hydrological Ecosystem Services as a potential policy instrument to improve IWRM operationalization at the management level. As a first step, a theoretical conceptualization of the state of the art of Payments for Hydrological Ecosystem Services and their implementation is carried out. Subsequently, their potential to improve IWRM operationalization at the management level is analyzed in a practical application context.

The theoretical argumentation of Chapters 4 and 5 forms the basis for a more specific empirical analysis of the PHES implementation, using a case study in Nicaragua as an example Chapter 6. In order to identify the specific IWRM implementation gaps and operationalization constraints in Nicaragua, the national IWRM process of the country is described and analyzed. This is followed by a documentation and analysis of the implementation process of a local Payments for Hydrological Ecosystem Services scheme. At the end of this chapter, an analysis of the contribution of local Payment for Ecosystem Service schemes to the operationalization process of IWRM and its functional role as part of a broader policy mix is realized.

Chapter 7 discusses the general potential of Payments for Hydrological Ecosystem Services to contribute to operationalization of IWRM and assesses their specific role within the policy mix for IWRM implementation in Nicaragua. Moreover, recommendations for PHES implementation to foster the IWRM implementation process are presented. The dissertation concludes with additional final remarks.

An overview of the chapters of this dissertation and a brief summary of their content is presented in Table 1.1.

Chapter	Brief description of chapter content
Chapter 2: IWRM introduction, problem analysis and state of the art implementation	Introduction to the concept of IWRM, its historical evolution and general principles; general analysis of the international implementation process; documentation and analysis of special implementation problems in developing countries; state of the art scientific discourse on the concept; principal trajectories and implementation gaps of the concept
Chapter 3: Methodological conceptualization of the core problems of IWRM implementation, identification of implementation instruments requirements	Methodological conceptualization of the core problems of IWRM implementation as the basis for the identification of specific requirements on complementary implementation instruments, identification of general operationalization constraints in developing countries, comparison of contributions of different policy instruments to the operationalization of IWRM, identification of PHES as a potential instrument to improve IWRM operationalization in developing countries
Chapter 4: State of the art of the Ecosystem Services approach and valuation methods, potential to improve IWRM implementation at the governance level	Documentation of the state of the art of the Ecosystem Services approach and methods of valuation with focus on hydrological ecosystem services, analysis of potential contributions in addressing the core governance problems of fit and interplay in the context of IWRM implementation.
Chapter 5: Analysis of PHES as an instrument to improve IWRM operationalization at the management level.	Theoretical conceptualization of the state of the art of Payments for Hydrological Ecosystem Services and their implementation, analysis of their potential to improve IWRM operationalization at the management level.
Chapter 6: Case study analysis: contribution of the implementation process of PHES to the operationalization of IWRM in Nicaragua	Documentation and analysis of the national IWRM process in Nicaragua, identification of implementation gaps and operationalization constraints, documentation and analysis of the implementation process of a local PHES scheme, analysis of the contribution of local PHES schemes to the operationalization process of IWRM in the context of a policy mix.
Chapter 7: Conclusions	Assessment of the general potential of PHES to contribute to operationalization of IWRM, specific assessment of the role of PHES within the policy mix for IWRM implementation in Nicaragua. Concluding remarks on the main findings.

Table 1.1: General structure of the dissertation with its principal parts and content of corresponding chapters

2 Evolution and implementation of Integrated Water Resources Development and Management

The foundation of this dissertation constitutes a thorough analysis of the historical evolution of the concept of Integrated Water Resources Management (IWRM), the general principles it contains as well as its pathway towards implementation in practice. This analysis of the evolution of the concept and the status of its global implementation is followed by an analysis of the scientific discourse on the background of the limited implementation success. Finally, the chapter closes with the state of the art of the present IWRM trajectories towards improvements in implementation and identified implementation obstacles. The objective of this chapter is to provide a clear problem statement and analysis

2.1 Evolution of a management framework

In order to understand the evolution of the concept of Integrated Water Resources Management (IWRM) and the persisting challenges of its implementation it is necessary to look at the history of water management, including the main actors and focuses of action.

IWRM (and other similar terms such as Total Water Management, Comprehensive Water Management or just Integrated Water Management) have been developed as a response to the shortcomings of traditional supply-based water management and are broadly compatible approaches. The application of these approaches at the river basin level as management unit is referred to as Integrated River Basin Management (IRBM) (also called Integrated Catchment Management in some countries). Traditional water management used to focus on technical solutions, with individual and often unconnected as well as uncoordinated projects, designed and implemented by different agencies representing different sectors, e.g. agriculture, energy, transport etc. (cf., Jones et al., 2006). As a result of this traditional approach, the management of water became highly fragmented, with multiple, sometimes overlapping, administrative and management responsibilities. Additionally, the traditional approach has been characterized by focusing on supply *fixes*, with little regard paid either to managing demand or to minimizing adverse environmental and social impacts. Besides a segregation of management approaches between different sectors, traditional water management has typically also been fragmented spatially, with administrative and political boundaries determining decisions about water use (Jones et al., 2006).

In the context of the evolution of IWRM as an internationally accepted framework for the sustainable management of water resources, it is not quite appropriate to refer to it as a *new* approach to water resources management, because IWRM is rather a comprehensive incorporation of good management practices that have been applied for a while to different degrees at different places and times to address a globally prevailing water crisis. *Traditional* versus *integrated* or *improved* management approaches are probably better terms for a conceptual distinction. According to Grigg (1999), it is the balancing of different goals and views of interdependent players that distinguishes this *integrated* form of management from other forms of management practices. Therefore, Grigg introduces the terms *disintegrated* and *integrated* for a clearer conceptual distinction.

Accordingly, there have been several forerunners of IWRM in different countries where water management has been institutionalized in an advanced and integrated way over centuries. Rahaman and Varis (2005) note that in Valencia, Spain "[...] multi-stakeholder, participatory water tribunals have operated at least since the tenth century". The river basin as reference for water management was probably also first applied in Spain with the system of hydrographic confederations (confederaciones hidrográficas) in 1926 (Embid, 2003). In the 1930s the Tennessee Valley Authority (TVA) in the United States became the first institution of water resources development integrating several sectors (integration of flood control, pollution control, water supply and conservation) in order to secure the efficient use of water resources of a whole region based on rational comprehensive planning as a centralized top-down and expert-driven approach by a single central authority (White, 1957; Mitchell, 1990)[1]. Moreover, at the United Nations Scientific Conference on the Conservation and Utilization of Resources in 1950 agreement was reached on the comprehensive development of river basins as the most important method not only of conservation but of harnessing the vast power and water resources of the world's great rivers (United Nations Department of Economic Affairs, 1950). In 1956 the Secretary-General United Nations (UN) declared that river basin development was an essential feature of economic development. White (1957) highlights subsequently that the comprehensive river basin management approach "[...] focused on the full utilization of rivers, multi-purpose dams, and wider regional development planning".

Relative to the availability of water resources (supply side) and the degree of utilization of them (demand side), a gradual transition has been observed in the way water resources are managed, as a response to increasing scarcity

[1] According to Teclaff (1996) this idea of river basins as economic development regions was later exported to Asia and also South America (e.g. Cauca Valley Authority in Colombia, São Francisco Valley Authority in Brazil). Later on similar authorities were establish at the international level in order to coordinate the development of transboundary river basins (e.g. La Plata River, The Amazon River).

and competition among uses. Mar (1998), for example, describes this transition as different eras of water resources management - from an *exploitation* era of unlimited availability and technology-based provision at the lowest possible economic costs, over an era of *management* of optimization of increased competing uses (including new concerns for water quality and groundwater exploitation) to a third and final era of *protection*, when demand "[...] becomes so great that a small increase in demand can destroy the supply" (Mar, 1998) and parts of the resource base are protected from use. The allocation of environmental flows to protect the natural habitat are a typical action that delineates the protection era according to Mar (1998).

Thus, several elements of integrated management of water resources had already been implemented, when a more holistic view on water resources management, described as IWRM, began to develop in the 1970s with the United Nations Conference on Water held in Mar del Plata in 1977, when IWRM was recommended as an approach to incorporate the multiple competing uses of water resources. Less attention was paid in the 1980s to the topic of water resources, thus, the famous "Brundtland"-report from 1987 (WCED, 1987), for instance, did not deal with it at all, although it paved the way for the following conferences on sustainable development (*Earth Summits*) and promoted the general discourse on sustainable development which later on began to focus strongly on water as an important cross-cutting issue. Accordingly, Lenton and Muller (2009) claim that considering IWRM as the water element of a broader sustainable development approach is quite appropriate. The three key strategic objectives of IWRM (the *three E's*) that have to be balanced reflect the sustainability goal of the approach:

- Efficiency to make water resources go as far as possible;
- Equity, in the allocation of water across different social and economic groups;
- Environmental sustainability, to protect the water resources base and associated ecosystems (GWP, 2013a; Lenton and Muller, 2009).

The development of IWRM gained further momentum in the 1990s and at the beginning of the 20th century (cf., Savenije and Van der Zaag, 2008). Important milestones towards an international consensus on IWRM as a means to achieve sustainable water resources management were the Informal Consultation on Water Resources in Copenhagen in 1991, where the demand driven approach (as opposed to the traditional supply-led approach) and the subsidiarity principle (to manage water at the lowest appropriate level, calling for more decentralized decision-making and participation) were launched (DANIDA, 1991) and the United Nations Development Programme (UNDP) Symposium on Water Sector Capacity Building in Delft (also in 1991), where the essential role of capacity building was recognized and the concept of IWRM further elaborated (UNDP and IHE-Delft, 1991). At the International Conference on Water and Environment, an expert meeting held in 1992 in Dublin, IWRM was broadly adopted by all of the conference parties. Hence, the conference called in the Dublin Statement on Water and Sustainable Development:

> *"[...] for fundamental new approaches to the assessment, development and management of freshwater resources, which can only be brought about through political commitment and involvement from the highest levels of government to the smallest communities. Commitment will need to be backed by substantial and immediate investments, public awareness campaigns, legislative and institutional changes, technology development, and capacity building programmes. Underlying all these must be a greater recognition of the interdependence of all peoples, and of their place in the natural world" (ICWE, 1992).*

The conference report's recommendations for action at local, national and international levels are based on four guiding principles (so called *Dublin Principles*):

Principle No. 1:
: Fresh water is a finite and vulnerable resource, essential to sustain life, development and the environment.

Principle No. 2:
: Water development and management should be based on a participatory approach, involving users, planners and policy-makers at all levels.

Principle No. 3:
: Women play a central part in the provision, management and safeguarding of water.

Principle No. 4:
: Water has an economic value in all its competing uses and should be recognized as an economic good.

The Dublin Principles became an important input to the United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro in 1992, and are therefore often referred to as *Dublin-Rio Principles*, which culminated in the adoption of the Fresh Water Chapter (Chapter 18) of the Agenda 21. The Fresh Water Chapter deals with water resources under the title "Integrated Water Resources Development and Management" and states that "[...] Integrated water resources management is based on the perception of water as an integral part of the ecosystem, a natural resource and a social and economic good, whose quantity and quality determine the nature of its utilization" (UNCED, 1992). This definition of IWRM underlines the importance of water resources as an integral part of ecosystems and the bio-physical limitation of the resource. Moreover, the Dublin-Rio Principles place a strong emphasis on governance aspects by stressing the need for participation of all stakeholders, awareness raising and addressing competing uses of water.

2.1.1 Starting the implementation of IWRM - Refining the concept and goal setting

The period until 1992 can certainly be described as a *conceptual phase* for IWRM on the international political agenda. The Dublin and Rio conferences clearly mark a milestone for the international evolution of IWRM, with the Dublin-Rio Principles representing a first conceptual agreement upon the way how water resources ought to be sustainably managed. Rio's Agenda 21 already included a first international goal setting element with the recommendation to all states to begin to implement IWRM principles by the year 2000 (Petit and Baron, 2009). Furthermore, IWRM as a policy approach was assigned a central role for the achievement of international sustainability and development goals.

An early phase of implementation followed the conceptual phase from 1992 onward until 2002. During this decade important steps towards international implementation of IWRM were taken and also much effort was made to clarify the IWRM concept in (more) practical terms. After Dublin and Rio, with the call for integrated management, the high degree of fragmentation of the water sector in the international community, and in particular the UN family, became obvious (cf., Savenije and Van der Zaag, 2008). One important step in the process towards more coordination within the water sector of the international community, since there is no UN organization that deals specifically with water resources, has been the formation of the Global Water Partnership (GWP) and the World Water Council (WWC) in 1996. Both organizations aim to coordinate the implementation of IWRM principles and practices worldwide. By providing a platform to encourage debates and exchanges of experience, the WWC aims to reach a common strategic vision on water resources and water services management among all stakeholders in the international water community. An important instrument of the WWC is the World Water Forum taking place every three years. The GWP focuses on the implementation of IWRM concepts at the operational level and advises countries, especially developing and newly industrialized countries, in the process of IWRM establishment.

The establishment of the GWP and the WWC a few years after the international consensus on IWRM in 1992 reflects the awareness of the international community of the forthcoming implementation challenge posed by this ambitious water management paradigm. From 1998 onward the GWP published several background papers on IWRM focusing on how to implement the principles agreed on in Dublin and Rio. With its Background paper No. 4 (GWP, 2000) the GWP published a comprehensive description of the IWRM concept and its implementation process. The publication provides advice and guidance on how IWRM could be implemented under different conditions. Moreover, it contains the following definition:

> *"IWRM is a process, which promotes the coordinated development and management of water, land and related resources in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems" (GWP, 2000).*

This definition is broadly accepted and describes IWRM as a *process*, often also referred to as the establishment of a normative framework for sustainable water resources management. Moreover, the definition emphasizes that coordination in (infrastructure) development and management of water resources is required, and water as well as land and related resources have to be considered together. The IWRM process is further attributed to be an open, flexible process bringing together decision-makers across the various sectors that affect water resources, and bringing all stakeholders to the table to set policies and make balanced decisions in response to the water challenges faced. Finally, it highlights the sustainability goal of economic and social welfare without compromising the ecosystems that sustain it. Based on this definition, the IWRM concept explicitly challenges traditional, sectoral water management concepts and places emphasis on an integrated approach with more coordinated decision making across water use sectors as well as across temporal and spatial scales.

Although different organizations have found alternative IWRM definitions (see Huppert (2005) for a detailed discussion of IWRM definitions and their degree of intra- and intersectoral integration), the above quoted GWP definition, although neither unambiguous nor officially accredited, is broadly regarded as the most authoritative one in existence (cf., Snellen and Schrevel, 2004; Lankford et al., 2007; Medema, 2008).

The GWP Background paper No. 4 on IWRM was published because of a perceived "[...] need for a clarification and formulation of certain principles and recommendations within integrated water resources management - serving a general purpose of contributing to the implementation of IWRM, but also an internal purpose of establishing a common understanding within GWP and Technical Advisory Committee of the Global Water Partnership (TAC)" (GWP, 2000). Thus, the paper represents the "corporate view" of the TAC on integrated water resources management. Nevertheless, and taking into account the pioneer and widely accepted role of the GWP, the document provides several clarifications on the principles underlying IWRM, its definition and how to implement it in general terms.

Because the Dublin-Rio principles have found universal support among the international community as the guiding principles underpinning IWRM, the GWP provides in its background paper more detail regarding the meaning of each of the principles (see below). This can be interpreted as a reaction of several controversies, above all about the exact meaning of water as an economic good, about the principles.[2]

- Principle I: Water as a finite and vulnerable resource
  - **A holistic approach to management** is needed, recognizing all the characteristics of the hydrological cycle and its interaction with other natural resources and ecosystems, is needed. Water is required for different purposes, functions and services - holistic management involves consideration of the demands and threats placed on the resource.
  - **Resource yield has natural limits** as the hydrological cycle on average yields a fixed quantity of water per time period; the resource may be regarded as a natural capital asset, which needs to be maintained to ensure that the desired services it provides are sustained.
  - **Effects of human activities** have an impact on the productivity of the water resource, reducing the availability and quality of water by actions, e.g. mining of groundwater, polluting water and changing land use which alter flow regimes within surface water systems.
  - **Need for recognition of the linkages between upstream and downstream users of water**. This implies that dialogue or conflict resolution mechanisms are needed in order to reconcile the different needs.
  - **A holistic institutional approach** is needed to coordinate between the range of human activities which create the demands for water, determine land uses and generate water-borne waste products.
- Principle II: Participatory approach
  - **Real participation**; stakeholders need to be part of the decision-making process (directly at the local community level, democratically by elected or otherwise accountable agencies or spokespersons or through appropriate pricing in market processes) depending on the respective scale of water management decisions.
  - **Participation is more than consultation**; it requires that stakeholders at all levels of the social structure have an impact on decisions at different levels of water management.
  - **Achieving consensus**; a participatory approach is the only means to achieve long-lasting consensus and common agreement. Stakeholders have to recognize that the sustainability of the resource is a common problem and that all parties are going to have to sacrifice some desires for the common good.
  - **Creating participatory mechanisms and capacity**; governments have to make participation possible at all levels by creating mechanisms for stakeholder consultation at all spatial scales. However, the creation of consultative mechanisms as such will not lead to real participation. Creating participatory capacity involves awareness raising, confidence building and education, and also the provision of the economic resources needed to facilitate participation and the establishment of good and transparent sources of information.
  - **The lowest appropriate level**; participation is an instrument that can be used to pursue an appropriate balance between a top-down and a bottom-up approach to IWRM.
- Principle III: The important role of women
  - **Involvement of women in decision-making**; operational mechanisms and actions to ensure an equitable participation of women in IWRM are needed to ensure women's participation at all organizational levels.
  - **Women as water users**; need for mechanisms to increase women's access to decision-making since they are important water users.
  - **IWRM requires gender awareness**; need to ensure that the water sector as a whole is gender aware.
- Principle IV: Water as an economic good
  - **Water has a value as an economic good**; many past failures in water resources management are attributable to the fact that water has been - and still is - viewed as a free good, or at least that the full value of water has not been recognized.
  - **Value and charges are two different things**; to avoid confusion over this concept there is a need to distinguish clearly between valuing and charging for water. The value of water in alternative uses is important for the rational allocation of water as a scarce resource (using the 'opportunity cost' concept), whether by regulatory or economic means. Charging for water is applying an economic instrument to affect behavior

[2] The notion of the economic value of water rather than water as a universal right is highly contested by NGOs and human rights activists, because until the present day the Dublin Statement on Water and Sustainable Development is still the only binding UN document that makes a statement on the issue. However, in November 2002, the UN Committee on Economic, Social and Cultural Rights adopted General Comment No. 15 recognizing water not only as a limited natural resource and a public good but also as a human right. Adopting General Comment No. 15 is seen as a decisive step towards the recognition of water as a universal right, although the document has no legally binding power (Woodhouse, 2004). Finally, in 2010, through Resolution 64/292, the United Nations General Assembly explicitly recognized the human right to water and sanitation and acknowledged that clean drinking water and sanitation are essential to the realization of all human rights. Besides a detailed description of the quantity, quality and accessibility properties of water, it states that affordability of basic water needs requires costs that should not exceed 3 per cent of household income.

towards conservation and efficient water usage, to provide incentives for demand management, ensure cost recovery and to signal consumers' willingness to pay for additional investments in water services.

- **Useful water value and cost concepts**; the concepts of Total Economic Value / Costs taking into account use and non-use (intrinsic) values and costs has been found useful within IWRM.
- **The goal of full cost recovery** should be the goal for all water uses unless there are compelling reasons for not doing so.
- **Managing demand through economic instruments**; treating water as an economic good may help balance the supply and demand of water, thereby sustaining the flow of goods and services from this important natural asset.
- **Financial self-sufficiency versus water as a social good**; to be effective water resources management agencies and water utilities need to ensure that they have adequate resources to be financially independent of general revenues. At the same time direct subsidies for targeted groups (disadvantaged water users) may be required, but they need to be transparent (GWP, 2000).

Moreover, with its IWRM definition, the GWP background paper offers a way to tackle the challenge at the operational level to translate the agreed principles into concrete action. Acknowledging "[...] that the concept of IWRM is widely debated and an unambiguous definition of IWRM does not currently exist" (GWP, 2000) the paper goes further and highlights that IWRM implementation has to occur context-specific. By focusing on describing what *integration* means it offers guidance to further work in IWRM. An important statement then is that "integration is necessary but not sufficient" - integration *per se* cannot guarantee development of optimal strategies, plans and management schemes. One the one hand, the paper points out two basic categories where integration has to occur both within and between these:

- the natural system, with its critical importance for resource availability and quality, and
- the human system, which fundamentally determines the resource use, waste production and pollution of the resource, and which must also set the development priorities.

Integration within the natural system (based on GWP, 2000) incorporates:

- **Integration of freshwater management and coastal zone management**; reflecting the *continuum* of freshwater and coastal waters.
- **Integration of land and water management**; an integrated approach to the management of land and water takes as its departure point the hydrological cycle transporting water between the compartments air, soil, vegetation, surface and groundwater sources. As a result, land use developments and vegetation cover [...] influence the physical distribution and quality of water and must be considered in the overall planning and management of the water resources. [...] The promotion of catchment and river basin management confirms that these are logical planning units for IWRM from a natural system perspective. Catchment and basin level management is not only important as a means of integrating land use and water issues, but is also critical in managing the relationships between quantity and quality and between upstream and downstream water interests.
- **Green water** and **blue water**; a conceptual distinction can be made between water that is used directly for biomass production and 'lost' in evapotranspiration (*green water*) and water flowing in rivers and aquifers (*blue water*). Management of *green water* flows holds significant potential for water savings (crop per evaporated drop in rainfed and irrigated agriculture), increasing water use efficiency and the protection of vital ecosystems.
- **Integration of surface water and groundwater management**; the hydrological cycle also calls for integration between surface and groundwater management. The drop of water retained at the surface of a catchment may appear alternately as surface- and groundwater on its way downstream through the catchment.
- **Integration of quantity and quality in water resources management**. The deterioration of water quality reduces the usability of the resource for downstream stakeholders.
- **Integration of upstream and downstream water-related interests**; an integrated approach to water resources management entails identification of conflicts of interest between upstream and downstream stakeholders. The consumptive 'losses' upstream will reduce river flows. The pollution loads discharged upstream will degrade river water quality. Land use changes upstream may alter groundwater recharge and river flow seasonality. Flood control measures upstream may threaten flood-dependent livelihoods downstream. [...] Recognition of downstream vulnerability to upstream activities is imperative. Once again management involves both natural and human systems.

Integration within the human system (based on GWP, 2000):

- **Mainstreaming of water resources**; when it comes to analyzing human activities or service systems, virtually all aspects of integration involve an understanding of the natural system, its capacity, vulnerability and limits.
- **Cross-sectoral integration in national policy development**; water resources management systems must include cross-sectoral information exchange and co-ordination procedures, as well as techniques for the evaluation of individual projects with respect to their implications for the water resources in particular and society in general.

- **Macro-economic effects of water developments**; in situations where large amounts of capital are mobilized for water sector investments the macro-economic impacts are often quite large and deleterious to overall economic development.
- **Basic principles for integrated policy-making**; land use policy-makers must be informed about the water consequences downstream and the external costs and benefits imposed on the natural water system, consideration of the effect of water developments on water demand and other water uses, be aware of the trade-offs between short-term benefits and long-term costs and application of the precautionary principle, subsidiarity in water resources management is essential so that different tasks are undertaken at the lowest appropriate level.

And on the other hand it presents the following important complementary elements for the implementation of IWRM (see also Figure 2.1):

- the enabling environment - the general framework of national policies, legislation and regulations and information for water resources management stakeholders;
- the institutional roles and functions of the various administrative levels and stakeholders; and
- the management instruments, including operational instruments for effective regulation, monitoring and enforcement that enable the decision-makers to make informed choices between alternative actions. These choices need to be based on agreed policies, available resources, environmental impacts and the social and economic consequences (GWP, 2000).

Jønch-Clausen (2004) states, on behalf of the GWP, that “[...] implementing an IWRM process is in fact, a question of getting these *three pillars* right” and the development of national IWRM plans should be based on those.

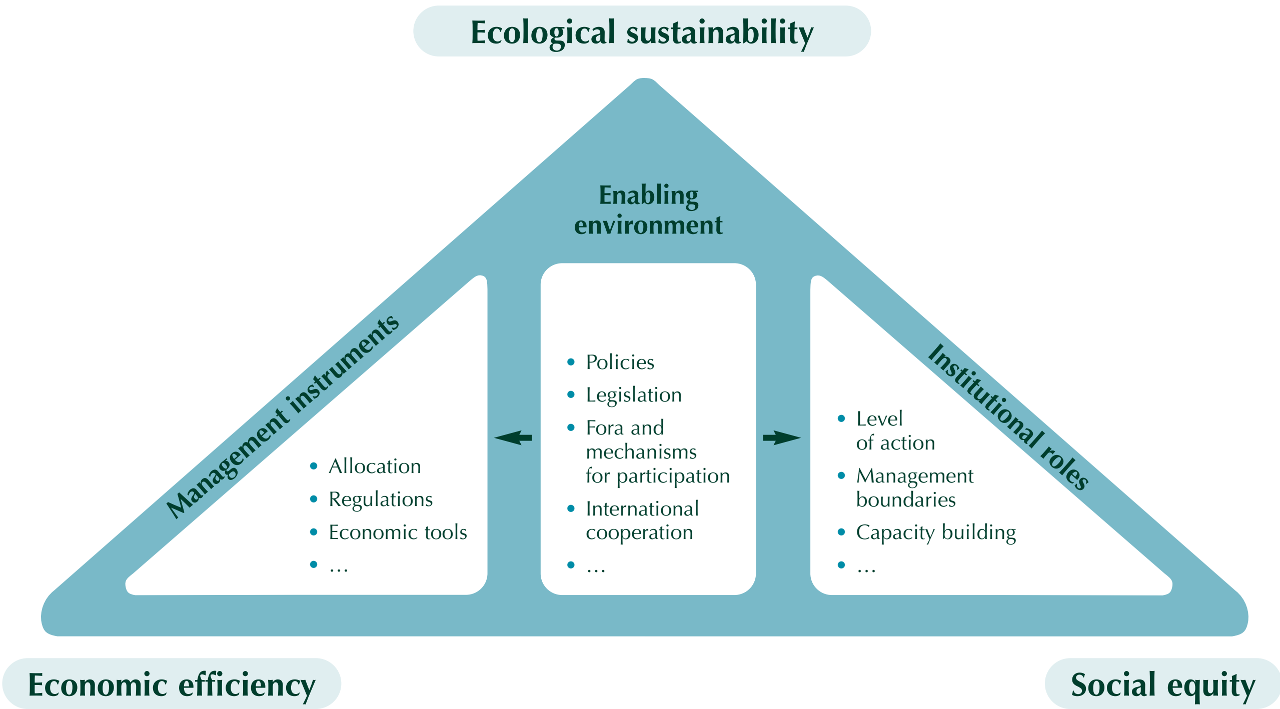

Figure 2.1: General framework for IWRM according to the GWP (2000)

These change areas are general in terms and may be applied on different political levels. At the national level general guidelines, institutional roles and management instruments need to be defined and established. On the one hand, this process inevitably has some top-down characteristics when creating a favorable enabling environment. By setting overall national policy and development goals, building an adequate legislative framework and by installing financing and incentive structures, pathways towards IWRM on the lower policy levels and institutional roles may be defined as well as management tools proposed or even prescribed. On the other hand, the change areas in terms of basin-wide cross-sectoral cooperation and conflict resolution, stakeholder involvement, public participation and recognition of the river basin as the basic unit for planning primarily represent a complementary bottom-up process.

Since the international agreement on IWRM in 1992, numerous reaffirmations and attempts to foster its implementation by the international community have followed. There have been reaffirmations to the IWRM concept, for instance, at the Second World Water Forum in The Hague, in 2000, with the formulation of the World Water Vision “[...] not just to end up the implementation of the Dublin Principles, but also to propose a comprehensive set of practical principles for implementation" (Cosgrove and Rijsberman, 2000). Furthermore, the GWP launched its IWRM ToolBox as an online-platform at the World Water Forum in The Hague (from 2001 on also available as a print version) intended as "[...] an information exchange platform where experiences are shared to help develop the body of knowledge which can enable all those engaged in water issues to work together to build water security and sustainable water for all" (GWP, 2012). The ToolBox builds on the above mentioned three IWRM implementation *pillars* enabling environment, institutional roles and management instruments. Within these three categories it offers 54 tools, i.e. guidelines, how to implement IWRM related to 193 cases of actual application of the tools in practice (GWP, 2013a). Moreover, the ToolBox contains 178 references in form of support documents, manuals, papers, and external IWRM knowledge databases. The tools part is the fixed part (revision possible, e.g. change from version 1 to version 2) and the case study and reference part is evolving dynamically with inputs from collaborators around the globe.

Jønch-Clausen and Fugl (2001) attempt to firm up the conceptual basis of IWRM as a reaction to the fact that "[...] IWRM has degenerated into one of these buzz-words that everybody uses but that means many different things to different people". Their paper is a summary of the GWP's background paper on IWRM with additional comments in order to improve the conceptual understanding of the concept. Most importantly, Jønch-Clausen and Fugl stress that IWRM is essentially a process and not a goal in itself and focus on what should be integrated. For them the integration within the human system is especially challenging. In their conclusion, Jønch-Clausen and Fugl emphasize that IWRM must not be interpreted as a universal blueprint for water resources management worldwide and therefore practical implementation must reflect local conditions of the natural and human system. The authors argue further that one should not be overwhelmed by the complexity of the IWRM concept. In practice, prevailing challenges should be at the center of IWRM implementation. However, getting the different actors to agree on an integrated approach and to act in a co-ordinated way may be much more difficult (Jønch-Clausen and Fugl, 2001).

In addition to the promotion of IWRM through the WWC and the GWP, the international community agreed on further measures to confront the difficulties in IWRM implementation. Based on the stated need for regular, global assessments on the status of freshwater resource at the Sixth Session of the Commission on Sustainable Development (CSD) in 1998 the member organizations of UN-Water founded the World Water Assessment Programme (WWAP) in 2000 to coordinate the production of the triennial UN World Water Development Report (WWDR) in conjunction with the World Water Forum. The WWAP's objective is to report on the status of global freshwater resources, changing management challenges and the progress achieved in reaching the Millennium Development Goals related to water. The first World Water Development Report was released in 2003.

In 2000 at the Millennium Summit of the United Nations the concept of IWRM was strongly related to the achievement of the MDG. Finally, at the World Summit on Sustainable Development in Johannesburg, in 2002, the UN adopted the establishment of national IWRM plans as an instrument to achieve the MDG. As a follow-up to the MDG based on the Millennium Declaration adopted by the heads of State in 2000 it was further agreed at the Summit in Johannesburg in 2002, through the Johannesburg Plan of Implementation (JPI), to 'develop integrated water resources management and water efficiency plans by 2005, with support to developing countries, through actions at all levels". The former deadline for IWRM implementation of IWRM plans was extended to 2005 at the 2002 Johannesburg conference (Article 18 of the summit's declaration). This extension was relayed by numerous international institutions such as the UNDP, the United Nations Environmental Programme (UNEP), the World Bank, the WWC and the GWP. UN-Water was founded in 2003 to foster greater co-operation and information sharing among UN entities and other relevant stakeholders.

The first phase of IWRM implementation revealed that IWRM implementation is a difficult and complex task. Besides first successful steps towards IWRM implementation, for instance in the EU, Canada, South Africa, and Australia, there has been little progress on implementation in other parts of the world. While important groundwork on the IWRM concept, of what it implies and how it may be implemented could be realized, the need for much more investigation on success factors and the overcoming of implementation obstacles became apparent. The need to know more about the global status of IWRM implementation and to foster the experience exchange process among practitioners is reflected in the establishment of the World Water Assessment Programme and the GWP IWRM ToolBox launch at the end of the first implementation phase.

2.1.2 Assessing the global status of IWRM implementation - Progress, obstacles and recommendations

During the period from 1992 to 2002 IWRM gained even more importance within the international community. At the Johannesburg Summit IWRM obtained the status of a means to achieve the MDGs and the international community agreed on the development of national IWRM plans by 2005. With the establishment of the GWP and the WWC, communication and experience exchange of the IWRM concept was fostered. Especially in developing countries the GWP has assisted, as in the case of South Africa, in IWRM implementation and the GWP ToolBox provided important guidance for practical steps towards IWRM.

The subsequent period of IWRM development from 2002 onward can be described as a second phase of implementation. This period reveals a shift towards a stronger focus on the implementation progress of IWRM in practice. During this third period several actions to improve implementation, e.g. through capacity building and promotion of extensive experience exchanges, as well as to monitor and assess the IWRM implementation process were realized. The importance of capacity building can be seen in the formation of the Cap-Net, a nonprofit international capacity building network for Integrated Water Resource Management(IWRM) initiated by UNDP.

Since the Second World Water Forum in 2000 the topic of water governance has become increasingly prominent and is now regarded as one of the most important issues to achieve IWRM implementation (Rogers et al., 2003; Rogers and Hall, 2003; Kidd and Shaw, 2007). A dialogue on effective water governance was undertaken by GWP in partnership with the UNDP and the International Council for Local Environment Initiatives (ICLEI) in 2002 in order to gain more experience on what makes water governance more effective (Rogers et al., 2003). A key conclusion from this dialogue was that water governance cannot be isolated from development in society at large. In conclusion, the dialogue report recommends that water governance should build on existing governance arrangements wherever possible, capitalize on opportunities to improve coordination and redefine roles and responsibilities, and develop capacity among individuals and institutions in order to govern water resources more wisely (Kidd and Shaw, 2007; Rogers et al., 2003). Additionally, a broad scientific dialog on IWRM has commenced, presenting critical views on the applicability of the concept and the struggles for its implementation.

As called for at the end of the first IWRM implementation phase, the development progress of national IWRM plans has been monitored using country surveys at repeating intervals. The first two surveys in 2003 and 2005 were carried out by the GWP and had mainly informal character. The surveys focused on the formulation process of national IWRM plans without regard to their respective implementation status.

The surveys carried out in 2007/2008 and 2011 by UN-Water became the basis of official status reports on the implementation of IWRM plans to the UN CSD whereas the 2012 "report [based on the survey realized in 2011] is more extensive, covering more countries and addressing the development, management and uses of water resources, as well as the possible outcomes and impacts of integrated approaches" (UN-Water, 2012b).

The survey results of 2003 documented - for the first time and on a broad analytical basis - that the IWRM implementation progress is slow and had to be monitored in order to identify its main obstacles. In order to carry out the survey, GWP regional contact persons were asked to provide a "[...] relative assessment of countries' maturity relating to the adoption of an IWRM approach. This assessment was to identify countries as having reached three different maturity levels (good progress, some steps, and initial stage) and to be qualified by short summary statements based on the survey data" (GWP, 2004). This first snapshot survey of the GWP in 2003 revealed that only about 10 % of the 108 surveyed countries (45 in Africa; 42 in Asia and the Pacific; 22 in Latin America) had made good progress in developing their IWRM and Water Efficiency plans, 50 % had taken some steps toward developing their plans, while the remaining 40 % were only in initial stages of the process (GWP, 2004).

The first UN World Water Development Report (WWDR) of 2003 confirms the progress towards IWRM implementation as being quite slow. Nevertheless, the report highlights "[...] encouraging trends in the needed reforms, and in three areas in particular: 1. recognition of the need for sound water governance and of certain required reforms of policy and institution, plus enforcement of laws and regulation, that are essential to sustainable water development; 2. reform of water institutions and policies is now taking place in many countries, but progress is slow and limited; 3. the IWRM approach is accepted in principle, but implementation is partial in both developed and developing countries" (WWAP, 2003).

In 2005 the GWP again conducted an informal survey, similar to the one in 2003. It focused on policies, laws, plans/strategies and other planning documents prepared by the 95 surveyed countries. The purpose of the survey was to capture the status of IWRM policies, laws and plans, but not to assess what is actually being implemented (GWP, 2006). The survey report summarizes the results as follows:

> *"The survey indicates that approximately 21 % of the countries have plans/strategies in place or well underway and a further 53 % have initiated a process for the formulation of an IWRM strategy/plan. Therefore, according to the definition provided by the MDG Task Force it can be concluded that about three-quarters of the countries surveyed have met the target of initiating a process for the development of national strategies/plans. In these countries the survey indicates that the IWRM approach appears to be well accepted as the way forward for better water resources management and use. The remaining 26 % have made only limited progress and in many cases have expressed a wish to move forward but need support in this process" (GWP, 2006).*

Moreover, the survey identified that "constraints to planning and implementation of IWRM include a lack of political will to foster needed policy changes and to allocate needed financial and other resources, as well as inadequate awareness of water issues, and inadequate institutional capacity" (GWP, 2006).

The initial surveys by the GWP as well as the first WWDR of 2003 document the broad acceptance of the IWRM concept among national governments, however, both assessments already highlight that a formal reform process based on IWRM-oriented policies, laws and plans is not enough to implement IWRM. While the formulation of IWRM plans, strategies and laws is described as slow and sometimes partial, the next step of actual implementation of IWRM is still absent at large.

Additional findings presents UN-Water (2006) presents additional findings in a report from the 4th World Water Forum on the subject of "Implementing Integrated Water Resources Management" from two global surveys, one being the GWP survey of 2005 and the other one a survey of the Japan Water Forum (JWF) (JWF, 2006), and six additional regional overviews that had collected and analyzed information on the progress of incorporating IWRM principles into national planning around the world. Despite "[...] that the basic principles of IWRM are being introduced into legislative and institutional reform, albeit in some cases slowly," the report highlights that more attention needs to be paid to implementation: "Moving from planning to the follow-up phase of concrete action appears to be a stumbling block for many countries" (UN-Water, 2006). Moreover, it concludes "[...] that despite the admirable progress made in initiating IWRM planning and establishing an enabling institutional environment for IWRM in many countries, the slow progress made after success in the initial stages indicates that the realization of the IWRM target set in Johannesburg may in fact take many years to achieve. The type and level of change required, a shift in mindset as well as operational approaches, demands widespread institutional as well as social change, at all levels" (UN-Water, 2006). Thus, the UN-Water report reaffirms the findings of the former implementation assessments. As a conclusion, the report identifies key areas for improvement: capacity enhancement, civil society involvement, international support and coordination, monitoring and indicator development and environmental sustainability.

In 2006, the WWAP once again published a World Water Development Report. This second report mirrors the global survey results of 2005 by stating that there is a significant gap between policy-making (e.g. through establishing formal IWRM plans and the development of new water legislation) and its implementation in practice, especially in developing countries. The report identifies serious gaps in developing countries between land and water use policies and governance. Furthermore, in spite of the need to localize water management, many governments "[...] fail to delegate adequate powers and resources to make it work. On the other hand local groups are often without access to information, are excluded from water decision-making, and thus lack a capacity to act" (WWAP, 2006). The report continues to highlight that "[...] there is, in reality, limited practical experience of how it [IWRM] can be implemented. In the overall context of IWRM, relevant challenges to and opportunities for an improved integration of land and water governance have, surprisingly, received little attention. It has proven difficult to integrate or coordinate land and water in a meaningful way, particularly for the rural and urban poor who have been socially and politically marginalized, and largely excluded from access to land, water resources management and related services" (WWAP, 2006). The report stresses further that recent achievements in the development of sophisticated water policies and plans (e.g. European Union Water Framework Directive or the IWRM process in South Africa) need to be balanced, however, by a recognition that policy changes at national levels have often been only imperfectly followed through to effective implementation (Zimbabwe is a recent example). The report sees "[...] a tendency to separate policy-making processes from implementation" (WWAP, 2006). Taking these statements into account, the WWDR provides a much deeper and much more detailed analysis of practical implementation obstacles than former assessments. Effective decentralization, integration of land and water management, coordination and participation at the local level are revealed as having a central role for successful IWRM implementation. Furthermore, it seems to be unclear how, i.e. through which (management) instruments, implementation can be successfully achieved (limited practical experience). Hence, the report points out major questions that still need to be worked out to put effective IWRM into practice. The report summarizes the following points explicitly linked to governance issues:

- Who is in charge of integration? Who implements integration?
- Who decides what interests should be reflected in IWRM plans and policies? How should policy processes be governed to ensure that relevant stakeholder interests are duly reflected?
- How should conflicting interests and disputes be resolved? What are the appropriate formal and informal institutions and conflict resolution mechanisms for efficient and equitable water decisions?
- Is there really a need to integrate all water issues?

Aside from highlighting the implementation gap of IWRM, the second WWDR presents several recommendations on how to overcome this gap: "There is no blueprint for improved governance. This suggests that specific solutions - the ideal solution - may be less relevant and emphasizes the importance of enabling processes and frameworks that can be applied to resolve certain issues in situations of economic or other constraints and in contexts of change, that is, 'second or third best' solutions" (WWAP, 2006). Thus, the report highlights clearly that the local context matters for practical implementation. Moreover, implementation should occur while addressing the most felt ecological and social constraints, even if this may promote second best solutions: "New management approaches will be based on regional cooperation principles, focusing on river basins and aquifer systems, with an emphasis on social needs and environmental sustainability. They will focus on interrelated natural resources problems, reduce potential points of friction and stress, and eliminate conflicting demands through risk management and vulnerability assessment. Classical legal tools and more informal approaches both have important roles to play in defusing conflict and developing cooperation" (WWAP, 2006). The key recommendation that "IWRM has to be tailored to prevailing socio-economic conditions", though, can encounter obstacles: - lack of proper coordination of management activities and appropriate management tools - inability to integrate water resources policies - institutional fragmentation - insufficiently trained or qualified manpower - shortfalls in funding - inadequate public awareness - limited involvement by communities, NGOs and the private sector (WWAP, 2006).

The second WWDR provides important findings on how to overcome the encountered implementation gap of IWRM. There appears to be a general agreement that, apart from the establishment of a (national) enabling environment and the assignment of institutional roles, for practical implementation the local context and the specific problem-setting in both its socio-economic and ecological dimension matter. Accordingly, blueprint solutions or all integrating solutions are not useful. In order to achieve practical implementation the report's recommendations make clear that more pragmatism seems more promising.

In 2008 UN-Water prepared a first official global status report on the implementation of IWRM for the 16th session of the Commission on Sustainable Development. According to UN-Water the report "[...] provides the best and most objective comprehensive overview of the current status of water resources management" (UN-Water, 2008) at this time. The report is based on survey results of questionnaires by the United Nations Department of Economic and Social Affairs (UN-DESA) and UNEP from 2007, covering 104 countries of which 77 are developing or countries in transition and 27 are developed, i.e. member states of the OECD and EU. The report explicitly addresses the need "[...] to assess the extent to which countries have been able to go beyond simply having plans in place to the stage of implementing those plans and the extent to which tangible outcomes have been forthcoming" (UN-Water, 2008). Earlier surveys by the GWP had an informal character and assessed only the establishment of IWRM plans without considering their implementation.

UN-Water's 2008 status report illustrates that developed countries make good progress in almost all aspects of IWRM implementation, and further improvement is mainly needed in public awareness and gender mainstreaming. In developing countries "there has been some recent improvement in the IWRM planning process at national level but much more needs to be done to implement the plans". This supports the findings of the WWDR and UN-Water of 2006 (WWAP, 2006; UN-Water, 2006). 17 of the 77 developing countries surveyed (22 %) have national IWRM plans in place and partially implemented; a further 2 countries (3 %) have these plans fully implemented.

Table 2.1 illustrates a selection of important national government actions (documents that guide and regulate the use, conservation and protection of a nation's water resources) towards IWRM in developing and transitional countries. The status report highlights that "[...] there are many illustrations of the tangible benefits of implementing plans that have adopted the IWRM approach. There are examples at the national and international levels; of particular significance are the examples at the community and provincial levels for it is at these levels that so many societal gains can be made" (UN-Water, 2008). As recommendations for improvements in IWRM implementation, the report highlights the importance of water management within natural hydrological units and the need for approaches to be tailored to the individual circumstance of country and local region. This implies again that different countries will need a set of actions suited to their particular needs and that time schedules for implementation will differ from country to country depending on specific country circumstances.

In 2009, the third World Water Development Report once more takes stock of IWRM implementation and recalls on the fact that "[...] implementing integrated water resources management is proving more difficult than envisioned" (UN-Water, 2009). Having said this, the report concludes that even in the case of adequate policies and laws being in place, development of water resources will not take place without adequate funding of infrastructure and the institutional and human capacity of the sector. Besides the need for sufficient funding, institutional and human capacity development, the report additionally recommends consultation with stakeholders and accountability in planning, implementation and management to build trust, Payments for Ecosystem Services (PES) as an incentive for improving water management efforts and for supporting sustainable ecosystems and water security (UN-Water, 2009). The report highlights the need to take increasing uncertainty in water management into account, improving monitoring of water quantity and quality to properly manage water resources and risk management are suggested as suitable solutions (UN-Water, 2009).

The issues of risk management and uncertainty in complex water management, caused by both socio-economic developments and climate change, are taken up again in the fourth World Water Development Report of 2012 (UN-Water, 2012a). As a response to prevailing challenges, the fourth report explicitly calls for an adaptive management approach towards IWRM. This implies that "as IWRM becomes more adaptive it will involve more multi-sector and multi-disciplinary collaboration. It will also be necessary to look beyond what is traditionally considered water management and link it with decisions made in other linked domains such as land management, agriculture, mining and energy" (UN-Water, 2012a). Hence, it requires a shift towards more emphasis on the management of processes and people as well as policies based on institutional reform, incentives and behavioral change (UN-Water, 2012a). The traditional command-and-control approach is regarded by the report as "less effective in many situations" compared to the promoted adaptive approach that calls for "learning to manage by managing to learn". The report explains how this has to be understood: "In a more

Country	IWRM related government action (strategy, plan, policy, law)
Eritrea	Integrated Water Resources Management and Water Efficiency Plan - Ministry of Land Water & Environment (draft 2007)
Egypt	National Water Resources Plan - Ministry of Water Resources and Irrigation (2004)
Botswana	IWRM Strategy and Action Plan - Ministry of Minerals, Energy and Water Resources (2006)
Burkina Faso	Decree No.2003-220: Action Plan for IWRM in Burkina Faso (PAGIRE) - Ministry of Agriculture, Hydraulics & Fishing Resources (2003) Burkina Faso Water Vision - Ministry of Agriculture, Hydraulics & Fishing Resources (2000) Water Law No.002-2001- Government of Burkina Faso (2001)
Costa Rica	National Strategy for Integrated Water Resources Management - Government of Costa Rica (2006) National IWRM Action Plan - Government of Costa Rica (2006) National Water Law (No. 14585) - Government of Costa Rica (draft 2006)
Honduras	IWRM Action Plan - Honduran Water Platform (2006)
Nicaragua	General Law on National Waters - Government of Nicaragua (2007) Environmental Action Plan - Ministry of Environment (1994) IWRM Action Plan - Ministry of Environment (1998)
Argentina	IWRM Roadmap - Sub-secretariat of Water Resources (2007)
Brazil	National Water Policy (Law No. 9433) - Government of Brazil (1997) National Water Resources Plan - Ministry of Environment, National Water Council (CNRH) National Water Agency (2007)
Philippines	Medium Term Philippine Development Plan (2004-2010) - Government of Philippines (2004) Clean Water Act - Government of Philippines (2004) Integrated Water Resources Management Plan Framework - National Water Resources Board (2007
Syria	National water Policy - Government of Syria National Water Law (No.31) - Government of Syria (2005) IWRM and Water Efficiency Plan - In place but partially implemented

Table 2.1: Evidence of adoption and use of the IWRM approach in different countries; based on (UN-Water, 2008)

adaptive approach towards IWRM, knowledge needs to be multi-disciplinary, based on an understanding of society and nature, and able to facilitate integrated approaches so that water institutions and management actors can absorb, adopt and implement new forms of management" (UN-Water, 2012a).

Besides the social dimension of the described shift towards an adaptive IWRM, it apparently has a strong ecological dimension as well. Thus, the report highlights the central role of ecosystems in sustaining the water cycle. This role needs to be understood in order to facilitate effective water management:

> *"An inclusive, holistic and participatory approach to water policy and management permits identification of the full range of ecosystem services involved, where the risks are, and who is vulnerable to them and why. The role of land cover (vegetation) and soil in reducing hydrological risk illustrates the need to rethink water storage in ecosystem terms. The use or restoration of ecosystem infrastructure to sustain or improve water quality is already a widespread practice with a proven track record. Using ecosystem infrastructure to manage risks associated with flooding is another area in which interest, practice and demonstrated feasibility are rapidly developing" (UN-Water, 2012a).*

At the implementation level, the report proposes a three-step process to identify opportunities for a proactive management of ecosystems in order to reduce uncertainty and manage risk:

1. Identify the water management objectives as opposed to focusing on infrastructure (e.g. objectives are water storage or clean water, not dams or treatment plants).
2. Explore what ecosystems offer in terms of meeting the identified management objective(s) (e.g. storing water, reducing pollution), including through their conservation and/or restoration.
3. Reduce the uncertainties and risks involved in decisions by considering all ecosystem services directly involved or potentially impacted by various management options. This includes valuing multiple co-benefits, and examining trade-offs between them to determine desirable courses of action.

Moreover, the report states that to develop and implement an effective water resources management programme ideally both a top-down and a bottom-up approach are incorporated. While "a top-down approach, being more strategic in orientation, offers a general framework within which a water management activity or programme can be developed and implemented. A bottom-up approach, being more operational in orientation, can provide an accurate picture of relevant

’on-the-ground’ water issues, needs and uncertainties experienced by a wide range of actors and stakeholders” (UN-Water, 2012a). Once again special importance is assigned to the local level for addressing water-related issues, “as this is closest to the point of actual impact, thereby facilitating acceptance of needed actions, provided that it is adequately positioned and has the capacity to effectively deal with the issues” (UN-Water, 2012a).

In 2012 UN-Water presents the most comprehensive global status report on the implementation of IWRM so far for the UN CSD meeting in the same year. While the main purpose of the UN-Water report of 2008 was to take stock of the development and implementation of IWRM and Water Efficiency Plans, from the Johannesburg Plan of Implementation, the purpose of the 2012 report is to focus on progress in the application of integrated approaches to the development, management and use of water resources (UN-Water, 2012b). Hence, the report addresses planning and implementation as well as the possible outcomes and impacts of integrated approaches.

Again this report sees progress in IWRM implementation, but also further need for improvements when it states that “while there is still a long way to go, progress towards the goal of sustainable water resources management is undoubtedly being made” (UN-Water, 2012b). As an example, the report points out that of the 130 countries included in the survey, 64 % have developed integrated water resources management plans and 34 % report an advanced stage of implementation. In low and medium Human Development Index (HDI) countries, however, progress appears to have slowed, or even regressed, since the last survey carried out in 2008. Thus, the report reaffirms that “much remains to be done to finance and implement plans in many low and medium HDI countries” (UN-Water, 2012b).

The status report uses the three key change areas enabling environment, institutional roles (i.e. governance and institutional frameworks) and management instruments, developed by the GWP (see also Section 2.1), to assess of the IWRM implementation status. According to the survey results presented in the report, most of all progress has been best in the two first mentioned areas: Since 1992, 80 % of countries have embarked on reforms to improve the enabling environment for water resources management based on the application of integrated approaches as stated in Agenda 21 and affirmed in the Johannesburg Plan of Implementation. Institutional reforms have been undertaken in many countries and correlate well with countries implementing legal and policy reforms (enabling environment). Nevertheless, translating policy and legal changes into implementation is a slow process and only a minority of countries indicate progress with stakeholder participation in their institutional reforms. The most common constraints to the development of appropriate institutional arrangements reported are related to clarity of mandates, cross-sector coordination, capacity, and participation as well as general awareness of the concept.

Through the application of management instruments on the ground, water policies and laws are put into practice. Thus, the effectiveness of the policy and the law is largely seen from the effectiveness of its management instruments. The report reveals that despite success stories (e.g. Uganda, Brazil, Australia) it seems that integrated approaches do not arise by decree, but from mutual trust, appropriate mechanisms and gradual acknowledgment of the benefits (UN-Water, 2012b). When applying management instruments in terms of efforts to manage water resources based on the river basin problems of spatial fit often arise because existing political-administrative territories often have different boundaries (UN-Water, 2012b). Furthermore, the report concludes that “even when mandates are relatively clear coordination and cooperation between management organizations can still be a challenge, as reported by almost one quarter of countries reporting on constraints, including Cambodia, Greece, Uganda and Panama” (UN-Water, 2012b). Fragmented approaches to water resources management are acknowledged as a general cause for the perceived lack of coordination and cooperation that countries reported. Besides this institutional fragmentation there is also a lack of capacity resulting in the inability to regulate and enforce laws and policies (UN-Water, 2012b). Moreover, inadequate participation and awareness of decision makers, users and other key stakeholders were noted by almost one third of the countries reporting on management constraints (UN-Water, 2012b).

The UN-Water report of 2012 also indicates - in the majority of countries - a high degree of adoption of management approaches at the level of the basin or sub-basin, as recommended in Agenda 21 twenty years ago. Around half of the countries being at an advanced stage of implementation of this river basin approach. Besides these decentralized management structures, however, “[. . .] the country response is mixed and in many cases unclear with regard to stakeholder participation” (UN-Water, 2012b). Although, basin management structures often provide a mechanism and means for good coordination and integration to take place, the country responses summarized in the report suggest that formal structures are not enough; thus, coordination also requires trust and willingness to share information and resources (UN-Water, 2012b). Finally, the report concludes that “it is evident from the survey that a truly integrated approach is a long term process that requires on-going political commitment. There is clearly a move to follow through the legislative and policy changes with action on the ground but more effort and support is needed to operationalize improvements to water resources management” (UN-Water, 2012b).

With 104 countries having embarked on IWRM implementation it has become a strategic instrument for sustainable water management around the globe. This strategic importance was reaffirmed last year at the United Nations Conference on Sustainable Development (Rio+20) in Rio de Janeiro as a political commitment:

"We reaffirm the commitments made in the Johannesburg Plan of Implementation and Millennium Declaration regarding halving by 2015 the proportion of people without access to safe drinking water and basic sanitation and the development of integrated water resource management and water efficiency plans, ensuring sustainable water use. We commit to the progressive realization of access to safe and affordable drinking water and basic sanitation for all, as necessary for poverty eradication and to protect human health, and to significantly improve the implementation of integrated water resource management at all levels as appropriate. In this regard, we reiterate these commitments in particular for developing countries through the mobilization of resources from all sources, capacity building and technology transfer" (UN, 2012, Outcome 120).

Both, the survey results on IWRM implementation and the WWDRs prove that progress is made, however, most of all by creating an enabling environment through new laws and integrated policies and assigning institutional roles at the highest (national) level. In contrast, the process to operationalize IWRM on the ground through the employment of integrated management tools often remains an exception. Much has been learned from successful implementations and some recommendation can be derived. An important insight has been that much more attention needs to be paid to specific contexts and problem-settings. To successfully implement IWRM at the local level, prevailing problems have to be addressed first and incentives must be created for stakeholders to engage, in order to achieve coordination. Recently, calls for a more adaptive approach towards IWRM implementation as well as combining socio-economic and ecological goals have become louder (UN-Water, 2012a,b).

The following section reflects on the scientific discourse on the concept of IWRM. This discourse has intensified since the end of the 1990s and paralleled the international political dialog.

2.2 Scientific discourse on IWRM - Criticism, obstacles for implementation and recommendations

The assessment effort on implementation monitoring by the international organizations in charge (GWP, UNEP, UNESCO, UN-Water) is complemented by an extensive scientific discussion among researchers and practitioners. Many authors see the implementation outcomes of the IWRM progress critically and draw lessons from implementation practice so far. There seems to be a general agreement on the principles of IWRM and the normative character of IWRM, but, as in the case of the concept of sustainability as well, there is much controversy on what IWRM implies in its practical application.

First reflections about the emerging global paradigm of IWRM came up in the scientific discourse at the end of the 1990s and the beginning of the 20th century. Grigg (1999), for instance, attempts "[...] to add clarity to the popular but poorly-understood concept of integrated water management" by reflecting on the need for and purpose of integration and by providing suggestions about implementing improved approaches to integration. He questions the usefulness of the "new" concept of IWRM for a general application: "The goals of integrated water management are certainly important, but it is not clear that the term is operable at practical levels. In any case, we need a process to determine when such an approach is required and to get the commitments needed from the appropriate participants. Water management organizations may need to be convinced to participate, especially when their investments go for regional benefits, not only their own direct needs" (Grigg, 1999).

Furthermore, Grigg asks in his paper *"Who should lead and who should pay?"*, thus, referring to who should assume institutional and financial responsibility for an integrated approach. He sees principal "barriers to integration" in a lack of congruence of political and problem boundaries, disincentives to cooperation, and low perceived need for integration. Hence, according to Grigg "this leads to the question: why is an integrated approach necessary in the first place? The case for the integrated approach is based on perceived benefits; thus the reason to accept leadership is to make integration work, once it has been shown to be necessary". In 1999, shortly before the nowadays most recognized IWRM definition of the GWP was published, Grigg questions further "whether the term *integrated water management* is the best one to use. Is the term operable, or should other words, such as *coordinated, regional* or *cooperative* be used?". Grigg believes that the main benefit of using integrated water management as a paradigm is its focus on the blending of viewpoints, and he offers this definition: "Integrated water resources management is a framework for planning, organizing and controlling water systems to balance all relevant views and goals of stakeholders". Having established a clear term the next step is to decide whether resolving a certain management problem requires a joint rather than a separate action: "In other words, rather than solve problems in an age of complexity, recognizing and understanding them becomes more of an issue. When it has been shown that joint action is required, getting commitments from the players to commit to the joint endeavor is the next step. Then, finding and validating the leadership is required, and negotiating the payment arrangements". Hence, Grigg argues for a more problem-oriented and context-specific rationale for integration.

Moriarty et al. (2000) address the practical implications of IWRM and acknowledge that "while at the international level agreements are signed and consensus reached about IWRM, at the local level, and within water sub-sectors there continues to be much confusion as to what exactly the new paradigm implies, and how it should be addressed". Their paper outlines a methodology based on an interpretation of the Dublin Principles that helps to initiate a process of inclusion of IWRM principles within drinking water and sanitation projects. The papers of Grigg and Moriarty et al. evidence that questions about the operationalization of IWRM, especially in the context of achieving integration, have been raised early on.

In 2004, Biswas published his critical paper *Integrated water resources management: a reassessment* asserting many problems, both in concept and implementation, of IWRM as defined by the GWP. He argues that there has been little impact of IWRM in practice, which he attributes, at least in part, to the *amorphous* character of the concept and absence of guidelines to achieve its vision. Moreover, he questions if "[...] it is possible for a single paradigm of integrated water resources management to encompass all countries, or even regions, with diverse physical, economic, social, cultural, and legal conditions?".

Similarly, Rahaman and Varis (2005) refer to IWRM as "[...] the current buzzword of water resources development, [and] future challenges remain in reducing the gap between theoretically agreed policies and implementation". Rahaman and Varis argue further that the integration of different related sectors is very challenging and "[...] overly general or universal policies and guidelines for implementing IWRM may become counterproductive" since problems and solutions of IWRM implementation differ. Thus, Rahaman and Varis resume that "three decades of conferences have resulted in many commitments to IWRM that, unfortunately, were often not implemented".

Van der Zaag (2005) offers a way out of the practical implementation dilemma by regarding IWRM more as a framework of possible management options than as a process. He concludes that "IWRM is a relevant, yet elusive and fuzzy concept", nevertheless, he regards IWRM as a must and not as an option. Van der Zaag discusses three general obstacles of IWRM implementation: the institutional dimension, decision-processes, and upstream-downstream linkages. Referring to the institutional dimension, Van der Zaag sees the role of new water organizations as consultative bodies ensuring consistent developments throughout the catchment without necessarily having executive functions. Moreover, in a bid to avoid wasting valuable resources, existing institutions and management structures should be incorporated. Accordingly, decision-making should be based on consensus. Thus, Van der Zaag again argues in favor of more pragmatic and context-related interpretation and implementation of IWRM. Concerning the upstream-downstream linkage obstacle, he states that "IWRM in this context means designing institutional linkages that reciprocate and mirror the water flows. This will promote equity and co-operation and preclude conflict. Our focus should be on the equitable sharing of the benefits derived from that resource and not necessarily on sharing the water itself" (Van der Zaag, 2005). Although not explicitly, Van der Zaag, at this point, already cautiously indicates the importance of recognizing the linkage between institutional design and the natural system to be managed.

But still more authors point at the implementation problems of IWRM. Watson et al. (2007), for instance, highlight "[w]hile most water researchers and managers appear to be comfortable with the general idea that IWM [Integrated Water Management] is concerned with balanced, equitable and sustainable management of water, land and other natural resources, there is very little agreement regarding what this actually means in practice or how IWM initiatives should be designed, implemented and evaluated. While previous research has shown quite clearly that IWM cannot be achieved through traditional top-down, fragmented and technocratic organizational and administrative arrangements, a great deal of uncertainty and disagreement still exists regarding the precise institutional processes and mechanisms that are needed".

Merrey (2008) makes a similar point as Biswas (2004) by claiming that "[IWRM] is leading to paralysis rather than prioritized actions as water managers struggle to implement the full normative IWRM package in all its complexity". He regards IWRM as a systems paradigm that is critical to understand problems and limitations of single-factor solutions. However, especially in the context of water management problems in developing countries (Merrey refers to his experiences from South Africa) his argument is analogous to the former cited authors that "[...] converting IWRM into a set of normative principles that must be implemented regardless of whether they are contextually appropriate is causing serious delays in solving real problems" (Merrey, 2008).

The far-reaching ambitions of IWRM and the respective broadness of goals are identified also by Medema (2008) as "[...] a significant hindrance to the ability of practitioners to demonstrate utility as the yawning gap between theory and practice is both difficult to bridge and offers an uncertain reward for those who try". García (2008) similarly argues "[...] the concept [of IWRM], despite the efforts of many to clarify the issue, represents many things to many people and accepts many definitions". Molle (2008b) calls IWRM "the main ubiquitous nirvana concept" in the field of water which "has evolved from the correct perception that water management has been unintegrated, or fragmented: economic sectors and ministries have managed water independently while interventions in, and development of, water resources in upper catchments have taken place without adequate consideration of impacts on downstream areas; water quality issues have been often either disregarded or disconnected from quantity issues; groundwater has frequently been exploited without concern for its hydrological linkages with surface water (and vice versa), and land-water interactions have been overlooked; and last, ecosystems have been impaired and social equity often disregarded". Hence, Molle argues that due to the "wooly" nature of IWRM the concept obscures the political nature of natural resources management and is easily hijacked by groups seeking to legitimize their own agendas.

Other authors agree with Molle stating that IWRM implementation is hindered if its inherent political nature is not seriously recognized (Allan, 2003b; Wester and Warner, 2002; Swatuk, 2005). Accordingly, Allan (2003b) emphasizes the importance of the the political nature of water management ("integration is political and management is political") by proposing to change *IWRM* to *IWRAM* with 'A' for Allocation. Although IWRM is derived from international consensus, Allan argues that it is reshaped by local political imperatives (cf., Molle, 2008b). Additionally, he sees in this context limitations of focusing on river basins as the fundamental unit given that economies and societies transcend hydrological boundaries. Wester and Warner (2002) question the transition to sustainable water management through the policy prescription emphasized by the dominant water discourse to manage water on the basis of river basins. They base this questioning on a concern that the political dimension of river basin management has not received sufficient attention: "A major problem with river basin management is that its political dimension has been neglected, through the reification of 'natural' boundaries, the emphasis on 'neutral' planning and participation and the search for optimal management strategies ('win-win' solutions). From a political science perspective, it becomes apparent that, at heart, the delineation and maintenance of boundaries, the mobilization of interests and stakeholder representation, and the creation of basin-level decision-making arrangements are quintessentially political processes that revolve around matters of choice" (Wester and Warner, 2002). The fundamental importance of governance is underlined by Wester and Warner in stating "that water management institutions and policies are effects of political practices". Recognizing this importance implies a "discourse that describes instead of prescribes, that focuses on processes and outcomes instead of forms and functions, and that is informed by real world struggles" in order to better understand water management practices and processes of institutional change.

Merrey (2008) agrees on the fact that water management is a political issue when he criticizes that many technical water professionals regard integration only with reference to hydrological and ecological dimensions while leaving out the political dimension. In their paper of 2005, Merrey et al. claim that IWRM should be centered on human welfare (livelihood-centered) as a multi-level approach. Local stakeholders need to take part in the decision-making process and serve as sources of information. Jonker (2007) supports this livelihood-centered IWRM conceptualization by arguing "IWRM is a framework within which to manage peoples' activities in such a manner that it improves their livelihoods without disrupting the water cycle".

Moss (2010) summarizes the criticism on IWRM that has been expressed in the scientific community besides a broad political consensus on the concept as follows (exemplary sources added by the author):

- Concept too vague (Biswas, 2004; Rahaman and Varis, 2005; Watson et al., 2007; Merrey, 2008; Molle, 2008b)
- Overplays win-win situations, downplays trade-offs (Allan, 2003b; Mollinga et al., 2007; Merrey, 2008)
- Based on normative claims rather than sound science (Biswas, 2004)
- Process-oriented, but lacking measurable targets for goals (White, 1998; Savenije and Van der Zaag, 2000; Thomas and Durham, 2003; Van der Zaag, 2005)
- Designed primarily for developed country contexts (Jonker, 2007; Lankford and Hepworth, 2010; Butterworth et al., 2010)
- Tension between integrative approach around river basins and participatory approach around local communities (IWMI, 2007b)
- River basins not always most suitable units for water management (Biswas, 2004)
- Parallel structures of decision-making: river basin and political territories (Wester and Warner, 2002; IWMI, 2007b)

2.3 IWRM implementation in developing countries

The different international IWRM implementation surveys showed clearly that developing countries are encountering more problems in translating legislative and institutional reforms into practice in order to achieve operational results. In 2007 the International Water Management Institute (IWMI) reviewed the IWRM implementation process in developing countries over the past three decades, in an attempt to promote institutional reform in the water sector (IWMI, 2007b). The IWMI report concludes that the major reason for disappointing results is that a succession of narrowly-focused blueprint solutions has been promoted and imposed, often with strong donor support. Moreover, the report states, for instance, that contrary to the implementation of often recommended river basin organizations "countries would do well to consider placing more emphasis on developing, managing, and maintaining collaborative relationships for basin governance - building on existing organizations, customary practices, and administrative structures". Consequently, the IWMI (2007b) proposes "a structured, context-specific approach to negotiating and crafting effective institutions and realistic policies that recognize the inherently contentious and political nature of institutional transformation".

The IWMI criticizes that what usually gets passed-off "in the name of IWRM at the operational [...] level has largely tended to include a blue-print package including:

1. A national water policy;
2. A water law and regulatory framework;
3. Recognition of River Basin as the appropriate unit of water and land resources planning and management;

4. Treating water as an economic good; and
5. Participatory water resource management".

Thus, according to the IWMI the difficulties in these blueprint packages lie in their practical implementation: "Drafting new water laws is easy; enforcing them is not. Renaming regional water departments as basin organizations is easy; but managing water resources at basin level is not. Declaring water an economic good is simple; but using price mechanisms to direct water to high-value uses is proving complex. As a consequence, the so-called IWRM initiatives in developing country contexts have proved to be ineffective at best and counterproductive at worst" (IWMI, 2007a).

Furthermore, several researchers point at the difficulties of developing countries in achieving tangible results from IWRM implementation. Merrey (2008), for instance, argues in a similar way as the IWMI stating that "[...] IWRM fails as a guide to practical action, at least in developing countries". García (2008) also makes this point: "The major hurdles encountered in this effort relate to the emphasis that was placed on applying the IWRM concept at the constitutional and associative levels compared to the operational level [...]. There is a tendency to be heavy on diagnostics but weak in solutions and a gap still exists between papers and actions".

Among other authors, Shah et al. (2006) have criticized the way IWRM has been interpreted and implemented in many developing countries "as a relatively standard 'package' of reforms regardless of the context" (as cited in Butterworth et al., 2010). According to Shah et al. this 'package' "[...] consists of the development of a national water policy, the creation of a water law and corresponding regulatory framework, development of water resource and service pricing mechanisms, creation of water rights by instituting a system of water withdrawal permits, the recognition of the river basin as the unit of water planning and management and subsequent creation of river basin organisations and the promotion of participatory water resources management". This critique goes in the same directions as the one expressed by the IWMI (2007a).

Butterworth et al. (2010) summarize critiques of Mollinga (2006) and Warner et al. (2009) that IWRM implementation in developing and transitional countries "often gives the impression of being externally imposed or adopted to please donor". Mollinga (2006) bases his critical point of view on experiences from South Asia when he refers to IWRM as a "concept in search of a constituency" since the concepts implementation, in his opinion, "is clearly not locally rooted". More explicit is Warner et al. (2009) more explicitly criticize that the new Kazakh water law had been drafted by an external consultancy along the lines of the European Water Framework Directive and, hence, unrelated to the realities on the ground. For this reason its implementation is therefore largely ignored (Warner et al., 2009; Butterworth et al., 2010). These criticisms again indicate the problems of IWRM implementation, when not adopted to the specific context of a country and merely based on general prescriptions.

In their comparison of IWRM implementation in the EU and in developing countries, Beveridge and Monsees (2012) point out as well that "an inherent challenge specific to developing countries is the general mismatch between IWRM concept and practices and the needs and conditions on the ground". Furthermore, the authors argue that key components like the river basin focus, the establishment of modern water rights, and demand management are much more suited to developed countries, where the IWRM concept, according to the authors, has its roots. Another developing country-specific challenge revealed in comparison to the EU is "a marked lack of human, financial and institutional capacity, in combination with strong pressures from normally project-oriented, short-term-oriented, external donors for quick, visible results (leading to rushed implementation)" (Beveridge and Monsees, 2012). Moreover, according to Shah et al. (2006), uncritical imposition of "institutional models in vastly different socio-ecological contexts can be dysfunctional and even counter-productive". Furthermore, Shah et al. stress the need to take a broader view of institutional change instead of focusing only "on things that governments can do - make laws, set up regulatory organisations, turn over irrigation systems, specify property rights" (Shah et al., 2006). Hence, the authors argue that many small steps to build the foundation for IWRM at the local level can be taken by informal actors instead.

A further obstacle of implementation is related to the decentralization process promoted by the IWRM approach. There are serious constraints to decentralization in developing countries because of lack of financial and personal resources (cf. Beveridge and Monsees, 2012). Thus, Butterworth et al. (2010) emphasizes that "at the local level, catchment agencies in many developing countries may be expected to struggle to establish legitimacy and be effective given their limited capacities, at least in the short and medium run". Furthermore, when catchment management agencies are in place to realize decentralization, effective participation in form of shared decision-making is often limited to the human and financial resources available to these agencies. Therefore, according to Butterworth et al. (2010) "such agencies often lack the capacity to fulfill even basic functions". As a result, IWRM is even more prone to be composed merely of top-down approaches without complementary bottom-up measures in developing countries since actions at the national or larger river basin scale are not reinforced and complemented at other scales. However, Lenton and Muller (2009) also highlight that in developing countries "IWRM reforms have tended to focus on the higher levels of scale, on policy and legislation reforms at national level and the establishment of river basin organisations".

Butterworth et al. (2010) see accepting realities in contexts of IWRM implementation as more promising than imposing ideals and standard policy packages. Thus, building on "effective existing local arrangements is more likely to succeed than starting from scratch at the catchment level. As local water users cannot wait for river basin organisations to develop enough capacity to effectively penetrate to the local level, much day-to-day decision-making on water development and management issues will remain in the hands of users and communities (e.g. in large parts of sub-Saharan Africa and the Andes)" (Butterworth et al., 2010). Accordingly, Butterworth et al. (2010) sees "opportunities not just to build upon the existing infrastructure, but even more importantly, upon existing institutions that already have the experience, knowledge and systems needed to manage water effectively at the local level. Some of these institutions are already quite integrated or relatively holistic while they may face challenges in adapting to be relevant at higher scales".

A good summary of key challenges for IWRM implementation in developing countries identify from a literature review, including several of those discussed above (exemplary sources added by the author), is provided by Beveridge and Monsees (2012):

- General mismatch with needs and conditions in developing countries (Allan, 2003b; Saravanan et al., 2009; Butterworth et al., 2010)
- Development politics and IWRM (Swatuk, 2005)
- Institutional fit and interplay (Moss, 2001; Saravanan et al., 2009; Horlemann and Dombrowsky, 2011)
- Lack of sensitivity to traditional, informal institutions (Swatuk, 2005; Shah et al., 2006; Van Koppen et al., 2007)
- Participation, equity and accountability (Molle, 2008b; Saravanan et al., 2009; Butterworth et al., 2010)
- Lack of resources and rushed implementation (Swatuk, 2005)

Despite this summary of Beveridge and Monsees (2012), particular insights on IWRM implementation in developing countries provide the results of International Conference on Integrated Water Resource Management (IWRM) entitled: *Lessons from Implementation in Developing Countries* which took place from 10 to 12 March 2008 in Cape Town, South Africa. The conference "highlighted the need for a concerted shift away from the debates on definitions and towards identifying implementation mechanisms and approaches" (Anderson et al., 2008). In accordance with the findings of past global IWRM implementation status reports and WWDRs, the conference conclusions highlight "the importance of balancing the establishment of enabling environments, which includes legislation, policies and institutional structures, with smaller-scale projects that have more tangible, immediate benefits for the poor". This statement indicates clearly that the predominating top-down approaches of IWRM implementation require specific complementary bottom-up approaches. The conference summary stresses further that "over-emphasis on the policy and legislation component leaves little benefit to those on the ground and does little to effect real change or promote poverty reduction" (Anderson et al., 2008). But at the same time, ignoring the enabling environment can limit the sustainability of IWRM by hampering longer-term formalization of IWRM approaches. The conference concludes pragmatically that "perfect integration between all sectors, across the hydrological cycle and between all users is unlikely" (Anderson et al., 2008). Hence, instead of waiting for perfect integration from top-down it is more important to achieve tangible benefits on the ground. In order to be successful, the conference conclusions highlight that the IWRM process on the ground must assist the achievement of benefits including increased access to water services, socio-economic empowerment, protection of ecosystems, improvement in water quality and overall poverty reduction (Anderson et al., 2008).

The conference paper of Leendertse et al. (2008) criticizes also that as a result of dominant top-down implementation, with little stakeholder engagement, "institutional and legal changes will have little effect on the way water is used and managed, with few tangible improvements in water quality and ecosystem protection" as cited by Anderson et al. (2008). Furthermore, Leendertse et al. (2008) argue that environmental concerns are often least considered when water management policies and plans are being developed and often even more neglected when in implementation. This is especially noteworthy since the poor are highly dependent on the environment.

At a workshop of staff members from IWRM case-study projects in developing countries held in 1996, a self-assessment tool was developed, based on an extended version of the Dublin principles that the staff members felt were relevant to their projects. A number of guiding questions and indicators relevant to each principle were developed by the participants and together formed an assessment tool. Moriarty et al. (2000) provide a summary of their assessment findings on implementation obstacles related to each of the eight extended IWRM principles (see Table 2.2).

In 2009 the GWP responds to the critics and admits that "IWRM has gained wide acceptance in water policy circles as the best way to tackle these challenges but 17 years after the approach was endorsed by the Rio Earth Summit, the content and even the relevance of the concept continues to be debated" (GWP, 2009a). "Following periods of conceptualizing and advocating an integrated water resources management (IWRM) approach and of establishing locally-owned regional and country partnerships" the GWP seeks with the book *Integrated water resources management in practice: Better water management for development* "to support countries to improve water resources management, putting IWRM into practice to help countries towards growth and water security" (Lenton and Muller, 2009). It is a reaction to the criticisms that:

- IWRM calls for integration of all activities that use or impact water resources, and thus is unworkable in practice (Biswas, 2008).
- IWRM is too broadly defined to have any real meaning or value except as a buzzword (Molle, 2008b).

- IWRM presents an ideal of water management but little practical guidance on how this ideal can be achieved in the real world (Wester and Hirsch, 2007).

Key IWRM principles	Key findings on implementation problems
Water source and catchment conservation and protection are essential	Water source and catchment conservation are increasing, but the necessary frameworks to ensure communication and cooperation between sectors and levels are often lacking.
Water allocation should be agreed between stakeholders within a national framework	While conceptually widely accepted, stakeholder, and user involvement remains limited and conflict between competing uses and users is often glossed over. Stakeholder involvement frequently occurs at an information rather than a decision making level. Good, appropriately presented hydrological information is essential to informed decision making, but is seldom available.
Management needs to be taken care of at the lowest appropriate level	The lack of clear legal frameworks enshrining rights and responsibilities within the decentralization process often causes confusion. While community-based approaches are now accepted as the norm, the necessary underpinning capacity seldom exists in support agencies.
Capacity building is the key to sustainability	Proper monitoring of the effectiveness of capacity building programmes is essential to their success. Capacity building programmes frequently pay insufficient attention to the lower and intermediate levels within decentralized support agencies, thus, they remain unable to fulfill their role in facilitating user decision making.
Involvement of all stakeholders is required	[. . .] communities frequently remain uninterested in becoming involved in wider IWRM because of high transaction costs and lack of genuine decision making powers.
Efficient water use is essential and often an important ‘source’ in itself	Water use efficiency and demand management is gaining attention, however guidance is often lacking in how to integrate it into projects.
Water should be treated as having an economic and social value	The principle of paying for water is now widely accepted and many projects are introducing water user charges. However, the role of water as a social good needs to be kept in view and the rights of vulnerable groups protected.
Striking a gender balance is essential	A wider understanding of gender as encompassing other important aspects of community dynamics such as age, wealth, class, cast etc. is missing. Gender specific approaches concentrate solely on the degree of involvement of women.

Table 2.2: Key IWRM principles and respective key findings on implementation obstacles; based on (Moriarty et al., 2000)

Consequently, the book is intended “to hopefully put to rest the concerns of some that IWRM is an unrealistic and impractical approach“ (Lenton and Muller, 2009). It emphasizes “that pragmatic, incremental approaches, which take into account contextual realities, seem to have had the greatest chance of working in practice” (Lenton and Muller, 2009). Accordingly, the GWP reports on common pitfalls in putting IWRM into practice (GWP, 2009b):

- “When it has been applied as blueprint - as a checklist of actions - in a way that does not take into account specific problems to be solved and contextual realities, IWRM has not delivered concrete benefits. Even within countries there are often significant differences that shape water resources challenges and possible solutions.
- Trying to establish management relations between too many variables risks getting mired in complexity at the expense of effectiveness. When putting IWRM into practice it’s important to think strategically about where and to what degree coordination and new management instruments are necessary.
- Participation can stall processes, undermine development and impose heavy costs on participants if it is undertaken without clear objectives and time lines, informed stakeholders, and mechanisms for negotiation and conflict resolution”.

In order to avoid common IWRM implementation pitfalls the GWP provides five key lessons learned from past implementation struggles:

- “IWRM is not a one-size-fits-all prescription and cannot be applied as a checklist of actions. Pragmatic, sensibly sequenced institutional approaches that respond to contextual realities have the greatest chance of working in practice.
- Water resource planning and management must be linked to a country’s overall sustainable development strategy and public administration framework.
- Water management must ensure that the interests of the diverse stakeholders who use and impact water resources are taken into account.

- Approaches to water resources management will evolve as the pressures on the resource and social priorities change. The challenge is to develop institutions and infrastructure that can adapt to changing circumstances.
- While the river basin is an important and useful spatial scale at which to manage water, there are often cases where it is appropriate to work at smaller sub-basin scale or at a regional multi-basin level" (GWP, 2009b).

A principle concern of all the critics is the continuous and prevailing implementation gap. Moss (2010) identifies six generic implementation problems from literature review:

- Gap between policy-making at national / international levels and implementation on local level
- IWRM often applied as standard 'package' of reforms regardless of context (lack of fit)
- Inadequate consideration given to building on existing structures
- Little consideration of informal norms and customs of local water management
- In developing countries IWRM can appear externally imposed
- Stakeholder participation in IWRM often weak in practice

Besides the scientific discourse on the limitations of a normative IWRM concept and the identified problems of operationalization, in recent years a discourse has also developed on implementation and operationalization in practice, especially in developing countries with limited resources. The following section analyzes the present trajectory of the IWRM concept and the state of the art of recommendations to improve its implementation in practice.

2.4 Present IWRM trajectories - Refining an established concept to improve practical implementation

Today a large majority of the world's countries have adopted an IWRM approach to achieve sustainable water resources management. There is broad agreement on the multi-level approach of IWRM that begins at the national, i.e. international, level by including water issues in a broader development agendas down to the river basin level and subsidiary decision-making. Cross-sectoral integration, stakeholder involvement and management of water resources on the basis of river basins (IRBM as IWRM on the level of river basins) are accepted as basic principles for IWRM implementation. This has led to the development of new national water laws, policies and strategies at the national level based on the principles of IWRM, forming the necessary enabling environment for IWRM implementation. Subsequently, institutional reforms have been implemented to improve integrated policy making.

However, this development has been based basically on top-down, regulatory and command-and-control mechanisms. While this strategy has been successful in creating enabling environments and establishing institutional roles at national, or in the case of the EU also international, level, little progress has been made in implementing IWRM *on the ground*, i.e. on the regional to local level of river basins and sub-basins. Even in the EU, where the implementation of the WFD is demanded in a command-and-control regulatory manner from top-down, remain challenges in following the river basin approach and in achieving public participation. In developing countries, due to apparent resource constraints, these challenges are even greater.

These shortcomings and obstacles in IWRM implementation have been repeatedly acknowledged by the international community as a principle result of the several surveys on the global IWRM implementation status and WWDR (see section 2.1.2). Additionally, as the discussion of the above documents illustrates, the scientific community concludes with a similar problem diagnosis on IWRM implementation. There is general agreement that there are no *blueprints* or *one-size-fits-all* solutions to IWRM implementation according to the GWP, WWC, UN-Water, and the WWAP (among others) representing the main promoters of the concept. Researchers generally support this conclusion, thus, questioning the suitability of *standard packages* for IWRM implementation (see Sections 2.2 and 2.3).

Besides criticisms and doubts on IWRM implementation, several researchers and practitioners have tried to adapt the IWRM concepts in order to improve its implementation on the ground. This sections provides an overview on the main outcomes and conceptual outputs.

As early as 2000, Moriarty et al. introduced a *light IWRM* approach that concentrates on the application of general IWRM principles (an extended version of the Dublin Principles; see Table 2.2 in Section 2.2) within water sub-sectors. The approach is based on experience from a comparative study by Visscher et al. (1999) of the degree of integration of IWRM principles into water sector (water supply and sanitation) projects in eleven cases from seven, mostly developing, countries: Zambia, South Africa, Ghana, Nepal, Cambodia, India, and Colombia. In general, the study revealed that while IWRM principles are internationally accepted they are not yet truly applied to the evaluated projects. Moriarty et al. (2000), thus, stress "while many national governments are addressing the issue of IWRM through the development of legislative frameworks, movement towards practical application remains slow". Hence, the authors "believe that the implementation of *light IWRM* will facilitate the eventual implementation of *full IWRM* as and when the necessary enabling environment comes into existence". According to the authors, the *light IWRM* approach "aims to help staff of sub-sector organizations to identify how they can best incorporate the relevant IWRM principles into their own projects and systems, rather than worrying about more abstract policy or regulatory matters that are outside their ambit of influence" (Moriarty et al., 2000). Thus, the proposed *light* approach aims at IWRM implementation starting from sub-sectors through which the application of the identified principles will start networking to achieve *full* IWRM in the long run. Thus, Moriarty et al. propose a context-specific IWRM approach based on the specific interests and concerns of sub-sectors at the community level. Furthermore, the direct involvement of water users in decision-making is highlighted (Moriarty et al., 2000). In a pragmatic way, they assume that no one can be an expert in all aspects, hence, in the efforts of each sub-sector to try to implement IWRM principles, the need to involve and include others will quickly become clear and integration will occur step-wise as a logical result (Moriarty et al., 2000). In 2004, Moriarty et al. highlight this point again stating that "any improvement in coordination or planning of water resource development represents a step in the process, and in many cases local level agreement and capacity-building on better sharing and use will have greater impact than new national laws or international level treaties" (Moriarty et al., 2004). Especially in the context of developing nations, they point out, "IWRM must not be viewed as a body of complex legislation, or an expert control system in which, to be effective, all aspects of water resources supply and use are integrated into a complex centralised system under the control of one super-agency" (Moriarty et al., 2004). While regarding basin-level IWRM by representative bodies in which all stakeholders are fully and fairly represented as the target, or endpoint, for achieving IWRM, Moriarty et al. (2004) propose to apply *light* IWRM in situations where over-arching legal and institutional frameworks for river basin planning and allocation of water resources are either missing or ineffective as a pragmatic approach tailored to meet capacities and contexts. This is often the case in resource scarce countries.

Similarly, the World Bank (2003) pursues in its *Water Resources Sector Strategy* a *pragmatic but principled* approach that recognizes "that water resources management is intensely political and that reform requires the articulation of prioritized, sequenced, practical and patient interventions". Therefore, the Bank insists to pay more explicit attention in design and implementation with "solutions that have to be tailored to specific, widely varying circumstances and that the art of reform is in picking the low-hanging fruit first, not in making the best the enemy of the good" (World Bank, 2003). The World Bank clarifies that it is not abandoning the idea of integrated water resources management but it recognizes that "even the world's most developed countries are a long way from integrated water resources management, and progress has been slow and incremental. The goal of this strategy is not to dismiss the goal of integrated water management, but to define practical, implementable and therefore sequenced and prioritized actions that can lead to that end" (World Bank, 2003).

Moriarty et al. (2010) applied and further developed their *light* IWRM approach within the 4-year (2003-2007) *EMPOWERS* project which was mainly funded through the European Union's Regional MEDA Water Programme for Local Water Management. *EMPOWERS* was implemented in Egypt, Jordan, and Palestine. The *light* IWRM approach was further developed specifically for use at the intermediate and local levels (sub-national and sub-basin) focusing on a facilitated process of stakeholder dialogue for concerted action supported by a strategic planning framework (Moriarty et al., 2010). Thus, representing a counterpoint to the typical package of institutional, legislative and informational interventions usually recommended and typically implemented in form of a national water sector reform (referred to as *full* IWRM by the authors). Thus, according to the authors, "in contrast to the normal top-down IWRM package, light IWRM aims to be pragmatic, problem-focused and adaptive" (Moriarty et al., 2010). Two elements are noted as particularly important to achieve this: (1) "real decentralized decision making can only happen with decentralized financing" and (2) "local water governance needs to be nested within higher-level water governance structures to give real effect to the principle of subsidiarity and to allow for scale-related and cross-boundary (physical and societal) issues to be dealt with" (Moriarty et al., 2010).

Another IWRM-project funded by the EU was *LoGo Water* - Towards effective involvement of local government in Integrated Water Resources Management in river basins of the Southern African Development Community (SADC) region (2005-2008). This project promoted a bottom-up IWRM approach through the active involvement of local communities which is very similar to the one pursued by the *EMPOWERS* project (Sullivan et al., 2008). The main components of these and other projects with focus on community participation (e.g. *WHIRL*, *WATERCOURSE*) are working with community owned knowledge and existing institutions, focusing on capacity building, active learning, identifying risk, engaging with local stakeholders and empowering them to make decisions (Heath, 2010).

The before mentioned attempts to improve IWRM implementation in practice focused strongly on reducing the complexity of IWRM at the local level and adapting the concept to local needs. Thereby, local priorities and decision-making were tried to place stronger emphasis.

Consecutively, with *expedient IWRM* Lankford et al. (2007) introduce an adaptive framework for IWRM in developing countries "which focuses relentlessly on *problems* on the ground rather than on IWRM principles to be articulated". They propose the problems identified in a certain catchment and the ongoing iterative relationships with stakeholders in the catchment should be the starting point of water management actions rather than a comprehensive framework of IWRM (Lankford et al., 2007). The comprehensive, idealized IWRM approach implies a package of tools and practices, designed to match and accommodate the complex nature of the water management problem, providing water managers with a long list of activities to execute, many of them simultaneously. Lankford et al. (2007), based on research in Tanzania complemented by a literature review, conclude that the attempt to implement such a comprehensive approach diverts

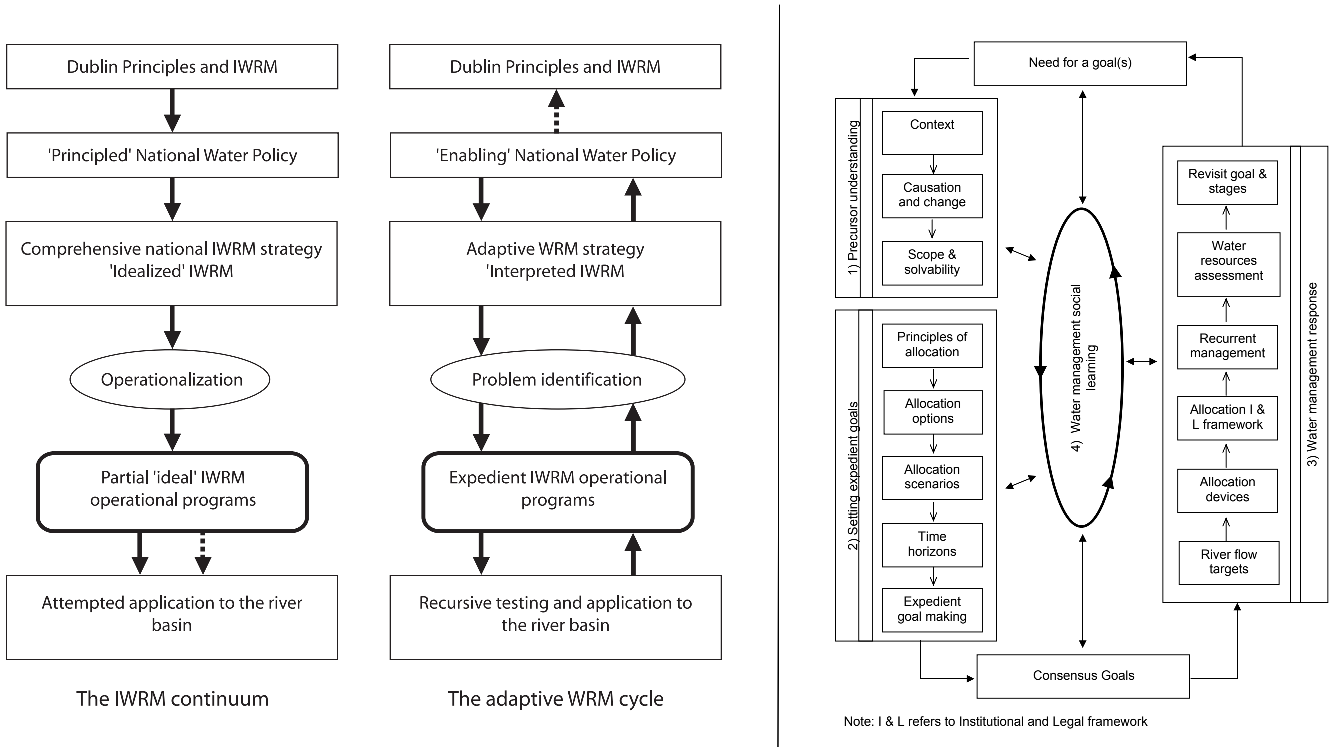

Figure 2.2: Normative IWRM and adaptive IWRM (on the left); Adaptive IWRM implementation cycle (on the right) from Lankford et al. (2007)

attention away from the most felt problems facing people in the basin and may even lead to inappropriate interventions. Therefore, a comprehensive approach is regarded by Lankford et al. (2007) as too complex, thus, requiring financial, human and institutional resources far beyond most developing countries' capacities (Lankford et al., 2007). Figure 2.2 illustrates the differences between the deployment of IWRM policy and operations of a partial comprehensive IWRM (on the right) implementation and an expedient (on the left) implementation. Thus, *expedient IWRM*, according to Lankford et al. (2007) "can be defined as 'advisable on practical rather than principled grounds', emphasizing a shift towards problem identification and solution, and away from the adoption of accepted norms - including the Dublin Principles" and has clear links to the concept of *adaptive management*.[3]

According to Lankford et al. (2007), IWRM implementation could be improved by focusing on "problems" rather than comprehensive achievement of all possible IWRM objectives (see Figure 2.2). Thus, Lankford et al. (2007) regard IWRM implementation as a "balancing act" between the theoretical ideal of IWRM and a strong orientation on prevailing problems. Mitchell (1990) similarly argues in favor of a differentiated implementation: "[...] at the strategic level, a comprehensive approach should be used to ensure that the widest possible perspective is maintained, but in contrast, a more focused approach is needed at the operational level where attention should be directed to a smaller number of issues that account for most of the problems" (Mitchell, 1990 as cited in Medema, 2008). While many countries have made good progress in taking a comprehensive approach at a strategic level by establishing legal enabling environments (e.g. national IWRM plans, new water laws) more focused approaches at the operational level are missing.

A structured, context-specific approach is also proposed by Merrey et al. (2007) in order "to negotiate and craft effective institutions and realistic policies that recognize the inherently contentious and political nature of institutional transformation". Mollinga et al. (2007) further develop and operationalize a proposal of Merrey et al. (2007) and advocate a *strategic action* approach to institutional reform (Merrey, 2008). In recognizing that water management is inherently political and embedded in a larger institutional context, a concrete problem setting is chosen as starting point by mapping what is necessary and possible and who are the actors involved. Mollinga et al. (2007) use the term *problemshed* to describe this alternative perspective as opposed to a purist watershed perspective. Therefore, a *problemshed* perspective implies identifying empirically the boundaries of a problem in contrast to pre-imposing a hydrological boundary. Thus, related parties to a problem are mapped and analyzed as an *issue network*. Merrey (2008) argues that these *issue networks* "will invariably include institutions and parties not contained within a specific basin or hydrological boundary". Hence, the focus is placed on "actually existing social relations in interaction and decision-making processes, and avoids the projection

[3] The MEA defines adaptive management as "systematic process for continually improving management policies and practices by learning from the outcomes of previously employed policies and practices" (MEA, 2005).

of ideal-type, normative models or assessments of social behaviour and decision-making" (Mollinga et al., 2007). This is the basis to create alliances around common goals and planning and negotiation of an implementation plan (Merrey, 2008).

Merrey (2008) notes that the *problemshed* approach of Mollinga et al. (2007) and the *expedient management* approach of Lankford et al. (2007) clearly converge. Both focus on specific issues and problems, begin with their prioritization and then identify options for solving the highest-priority problems by identifying the constellation of interested parties (actors and institutions) to negotiate a way forward (Merrey, 2008). Therefore, IWRM no longer constitutes a blueprint or objective in itself, it rather provides a systems context to understand the likely implications of specific solutions (cf., Merrey, 2008).

The discussion so far has revealed that attempts to foster implementation mainly from top-down by creating new institutions, forcing collaboration and drawing new administrative boundaries along river basins do not seem to be the best way to achieve integrated management results on the ground. An important insight from over two decades of experience with IWRM implementation among both practitioners and researchers is that IWRM implementation at the operational level has to occur context-related and orientated to specific prevailing problems on the ground - thus more *adaptive* - rather than being overly *comprehensive*. Hence, researchers are increasingly proposing *interpreted*, *expedient*, *light* or *adaptive* approaches starting from *bottom-up* in order to implement IWRM in a complementary manner to the prevailing top-down approaches. These, often pragmatic, proposals are based on the recognition that IWRM deals with complex systems involving the need to include risk and uncertainty in the management of water resources. Lankford et al. (2007); Mollinga et al. (2007); Merrey (2008); Moriarty et al. (2010) developed approaches to overcome the IWRM implementation gap, as a response to the strong political character of IWRM and local realities (people centered) based on empirical models. The authors do not provide a (theoretical) generalization of the systems characteristics to be managed nor of the management approach itself. Full IWRM implementation is regarded as a task far too complex to achieve in a normative manner in practice. Therefore, the solution, an adaptive / expedient approach, is developed to reduce complexity by building on prevailing problems, issues and participants. Hence, *adaptation* is interpreted as reducing complexity by using easier entry points and options specific to certain problem contexts (e.g. a *problemshed*). Responding to IWRM criticism from Biswas (2008); Molle (2008b); Wester and Hirsch (2007), the GWP (2009b) argued recently in a similar way that "pragmatic, sensibly sequenced institutional approaches that respond to contextual realities have the greatest chance of working in practice" and "there are often circumstances where it is appropriate to work at smaller sub-basin scale or at a regional multi-basin level".

Based on findings of Moriarty et al. (2000); Lankford et al. (2007); Moriarty et al. (2010), Butterworth et al. (2010) focus in their papers on *lighter* and more locally rooted approaches to water management. These approaches embrace "IWRM as a principle, they seek their application in an alternative manner: focusing at the more local level, as opposed to the river basin or national level; seeking integration from within sectors, as opposed to establishing inter-sectoral mechanisms; and building upon existing institutions and participation mechanisms, as opposed to establishing new multi-sectoral institutions" (Butterworth et al., 2010). A summary of IWRM criticisms and solutions/alternative entry points on how to address these with a lighter IWRM approach are presented (see Table 2.3). Butterworth et al. (2010) argue that these *lighter*, more pragmatic and context-adapted approaches, strategies and entry points are useful "in addition to the necessary but long-term policy reforms and river basin institution-building at higher levels". These approaches "address some of the scale problems in implementing IWRM, are attractive for their pragmatism rather than idealism, and make it easier (or unavoidable) to engage with people and politics" (Butterworth et al., 2010).

Despite all criticism, Butterworth et al. (2010) advise against discarding the concept of IWRM for its flaws. In summary, the authors argue to work towards the outcomes "that IWRM originally aimed to achieve though a better mix of complementary light and full approaches at different levels of scale that build upon local and sectoral realities" (Butterworth et al., 2010). Hence, the actual mix of light and full approaches should be determined through specific outcomes desired in a given location. According to the authors, this mix can only be developed through an adaptive approach which will certainly be subject to political response and local constraints (e.g. capacity and resources).

Beveridge and Monsees (2012) compare institutional challenges and politics of IWRM in developing countries and the EU and identify "similarities in the core challenges of achieving integration (e.g. [spatial and institutional] fit and interplay) and the requirements of 'good governance' (participation, etc.)". Institutional interplay and spatial fit (or misfit) dimensions of the integration challenge are inseparably linked to each other in terms of, "e.g. the definition of a 'competent authority', multi-level governance, cross-sector integration and spatial planning" (Beveridge and Monsees, 2012). Based on their literature review, Beveridge and Monsees (2012) stress "that in order to successfully implement integration in water resource management in both EU and developing countries, policies and projects need to be more sensitive to context-specific conditions". Consequently, Beveridge and Monsees (2012) argue that research "should be geared towards encouraging what Butterworth et al. (2010) have called a *lighter*, more *practical* IWRM: accepting realities in contexts of implementation, instead of imposing ideals and standard policy packages" (Beveridge and Monsees, 2012).

The context-specificity of IWRM implementation at the operational level that is repeatedly highlighted by several authors, is provided by two basic systems: the natural system (e.g through ecosystem properties) and the social or human

IWRM criticisms / problems	Solutions provided by light IWRM approaches
Vagueness of IWRM concept	IWRM should be considered more as a philosophy than as a ‘package of reforms’
No agreement on fundamental issues on what to integrate, how, by whom, or if integration is practically possible	IWRM principles should be built into projects and programs
IWRM is not sufficiently people-centered	Local laws and customary institutions should be an entry point for IWRM
IWRM does not incorporate adaptive management principles	Better linkages should be built with local government and its planning processes
Concept is unwieldy	IWRM should be built from bottom up
Packages of IWRM reforms do not include local IWRM	IWRM reforms need to build upon existing mechanisms for participation and organization of stakeholders, even if this means building upon ‘sectorality’, rather than a complete overhaul
River Basin Organizations may struggle to establish legitimacy	‘Light’ approaches that aim to apply IWRM principles at all stages of the project cycle are more likely to be good entry points
RBOs often lack the capacity to fulfill even basic functions	Supporting the existing local arrangements should be encouraged as a form of local IWRM in itself and is more likely to succeed than starting from scratch at the river basin level
IWRM activities ignore politics	Although local IWRM initiatives often have limited scope, they can still contribute to the development of IWRM at basin scale and serve as important entry points for applying the IWRM framework
Levels of participation in IWRM are low	Forging better links between the water, sanitation and hygiene (WASH) sub-sector and IWRM is another way to strengthen grassroots participation in IWRM

Table 2.3: Summary of common criticisms of IWRM and possible ways out by Butterworth et al. (2010)

system (e.g. through existing institutions, perceived problems, power relations, political organization etc.). Both systems are interdependent and place site-specific constraints on operational IWRM implementation. While Lankford et al., Merrey, Moriarty et al., Mollinga et al., and Butterworth et al. have stressed the importance of the human system context, other researchers call for adaptive approaches towards IWRM supporting their argumentation by referring to the theories of Social-Ecological System (SES).[4] This can be interpreted as further development of *light* approaches in taking into account the intrinsic interdependency of human and natural system contexts based on Social-Ecological System theory.

Pahl-Wostl and Sendzimir (2005), for instance, describe adaptive management as “seeking to increase the adaptive capacity of river basins based on an understanding of key factors that determine a basin’s vulnerability” (as cited in Lenton and Muller, 2009). According to Pahl-Wostl and Sendzimir, adaptive management is a concept that has its origins in ecosystem management (Holling, 1978) and “is based on the insight that the ability to predict future key drivers influencing an ecosystem, as well as system behaviour and responses, is inherently limited” (Pahl-Wostl and Sendzimir, 2005). Thus, adaptive management includes the ability to manage change by continually improving management policies and practices and by learning from the outcomes of implemented management strategies. The paper of Pahl-Wostl and Sendzimir (2005) discusses differences between traditional and integrated, adaptive approaches. The identified differences form the basis for subsequent analysis where Pahl-Wostl (2006) calls for a transition from current management regimes based on prediction and control to more adaptive regimes as a learning approach. Pahl-Wostl et al. (2010) develop this further stating more explicitly that “in terms of planning intervention therefore, the new paradigm implies a change from *command and control* to a more systemic approach rooted in the co-production of knowledge and acceptance of uncertainty“. The differences between the two approaches are summarized in Table 2.4.

In this context, the EU project HarmoniCOP (Harmonising Collaborative Planning; 2003-2005) was intended to increase the scientific understanding of social learning and public participation processes aimed at supporting the implementation of the WFD. To do so, the project’s conclusions propose to move towards new adaptive institutional regimes (Tàbara et al., 2005). Based on the experiences of the HarmoniCOP project, Pahl-Wostl et al. (2007) conclude that “collaborative governance is considered to be more appropriate for integrated and adaptive management regimes needed to cope with the complexity of social-ecological systems”.

4 Social-ecological systems are linked systems of people and nature. The term emphasizes that humans must be seen as a part of, not apart from, nature - that the delineation between social and ecological systems is artificial and arbitrary. Scholars have also used concepts like ‘coupled human-environment systems’, ‘ecosocial systems’ and ‘socioecological systems’ to illustrate the interplay between social and ecological systems.

Dimension	Prediction, control paradigm	Integrated, adaptive paradigm
Governance	Centralized, hierarchical, narrow stakeholder participation	Poly-centric, balance between bottom-up and top-down processes, broad stakeholder participation
Sectoral integration	Sectors separately analyzed, resulting in policy conflicts and emergent chronic problems	Cross-sectoral analysis identifies emergent problems and integrates policy implementation
Scale of analysis and operation	Transboundary problems emerge when river sub-basins are the exclusive scale of analysis and management	Transboundary issues addressed by multiple scales of analysis and management
Information management	Understanding fragmented by gaps and lack of integration of proprietary information sources	Comprehensive understanding achieved by open, shared information sources that fill gaps and facilitate integration
Infrastructure	Massive, centralized infrastructure, single sources of design and power delivery	Appropriate combination centralized and decentralized, diverse sources of design and power delivery
Finances and risk	Financial resources concentrated in structural protection (sunk costs)	Financial resources diversified using a broad set of private and public financial instruments
Dealing with uncertainties	Uncertainties perceived as undesirable sign of incomplete knowledge Emphasis on reducing uncertainties Influence of different perspectives largely ignored	Irreducible uncertainties accepted Emphasis on how to deal with uncertainties and robust strategies Influence of different perspectives explicitly acknowledged

Table 2.4: Two water management paradigms and their manifestation in characteristics of the water management regime (Pahl-Wostl, 2006; Pahl-Wostl et al., 2010)

Also Watson (2004) argues "[...] the current 'myth' [of inter-agency coordination] must be reformed and that a more powerful system-response capability founded on inter-organisational collaboration should be developed". He proposes in his paper that "[...] researchers and practitioners should shift their attention from co-ordination strategies to the design and implementation of more collaborative approaches for decision-making and problem solving". 'Wicked' or 'messy' management problems characterized by complexity, change, uncertainty and conflict, thus, cannot be solved through inter-agency cooperation alone, thus Watson (2004) calls for a reform of IWRM towards more collaborative and adaptive management approaches, taking into account the nature of river basins and society, i.e. the natural and human system context.

Timmerman et al. (2008) reviewed, in the context of the EU funded *NeWater* research project the state of the art in European research on integrated water resources management on the topics of participation, transboundary regimes, economics, vulnerability, climate change, advanced monitoring, spatial planning, and the social dimensions of water management. They conclude that the concept of adaptive water management should become a preferred direction for the future development of IWRM. Lenton and Muller (2009), on behalf of the GWP, as well regard the adaptive approach proposed by Pahl-Wostl and Sendzimir (2005) as "a useful focus".

Engle et al. (2011) observe the continuous merging of the concepts of IWRM and Adaptive Management. As both concepts have unique advantages, they also vary their adaptability and their level of institutional and stakeholder integration. Figure 2.3 illustrates the predominant characteristics of each concept and also includes the main characteristics of traditional command and control approaches, where both concepts depart from.

Pollard and Du Toit (2008) reflect on experiences of national IWRM implementation in South Africa by addressing the challenges of managing catchments comprised of linked social and ecological systems: "It is also widely recognised that the management of such systems requires an iterative, *learning-by-doing* approach that is reflexive in nature and builds learning into the next management cycle". In the opinion of the authors an adaptive management approach appears best suited to such conditions. In the context of complex systems, the role that water users play as a part of deriving management solutions is central to an adaptive management approach. Pollard and Du Toit (2008) argue that self-organization, identity and integration are all essential characteristics in order to build resilience in a catchment system.

Lankford and Hepworth (2010) introduce an alternative model of poly-centric water resources management (*bazaar*) in contrast to predominating centralized (*cathedral*) models. Lee (2003) defined poly-centric adaptive governance earlier in general terms as a "form of social coordination in which actions are coordinated voluntarily by individuals and organizations with self-organizing and self-enforcing capabilities" (as cited in Folke et al., 2005). According to Lankford and Hepworth (2010) "*poly-centric* river basin management, is institutionally, organisationally and geographically more decentralised, emphasising local, collective ownership and reference to locally agreed standards". The poly-centric model

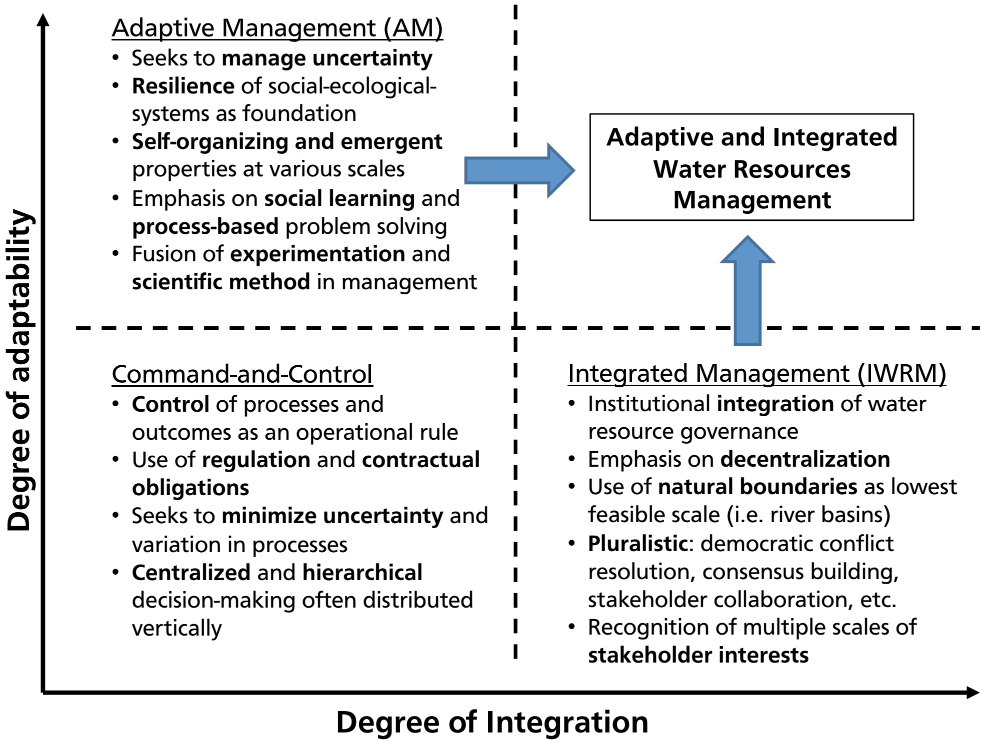

Figure 2.3: A simplified summary of major integrated water resources management (IWRM) and adaptive management (AM) principles (Engle et al., 2011)

proposed by the authors is constructed from the creation of appropriate managerial mosaics of nested sub-units within river basins rather than integrating it into a hierarchical whole (cf. Ostrom, 2009).

Accordingly, the "focus [is] on pursuing immediate and customised solutions to problems that subunit societies face including the provision of scientific services to support local policy-making and the fostering of local standard-setting and agreements on monitoring, rewards and penalties" (Lankford and Hepworth, 2010). These internally functioning sub-units, *holons*, subsequently nest together to achieve the basin-wide performance (see Figure 2.4). Uphoff (1996) claims that *holons* as introduced by Koestler (1967), may be considered as *whole* in themselves and at the same time *parts* of larger wholes. Hence, certain aspects of the behavior of a *holon* may be confined entirely to that *holon*, yet systems (i.e. *holons*) at any level are constrained and controlled by the levels above and below, thus, can be considered as nested (Stephens and Hess, 1999).

Pahl-Wostl and Kranz (2010) encounter some empirical evidence to support the hypothesis that poly-centric systems with effective stakeholder participation are more adaptive than either centralized or fragmented governance systems (Pahl-Wostl, 2009). Moreover, to be effective in increasing governance performance, poly-centric decentralization approaches should be combined with effective vertical integration and cross-level interactions (Pahl-Wostl and Kranz, 2010). Public participation and the involvement of stakeholders proved to be important for achieving effective vertical integration and cross-level interaction (Pahl-Wostl and Kranz, 2010). According to Pahl-Wostl et al. (2012), adaptive management with a poly-centric structure "creates possibilities of responding at different spatial scales as well as dealing with heterogeneity in impacts and capacities among different places or sub-basins". Due to the distribution of authority in a poly-centric regime, place-specific responses to heterogeneity and uncertainties are easier to achieve than in a centralized system. Pahl-Wostl et al. (2012) argue that past emphasis on legal frameworks and management plans in water governance reform was quite futile in countries with limited statehood where formal institutions are not effective. Thus, support of bottom-up informal processes to develop civil society and local governance capacity instead of intervention at the national level may be more promising (Pahl-Wostl et al., 2012).

Another group of authors also apply the theory of Social-Ecological System to improve IWRM implementation, but draw their attentions more towards the implications on how to deal with the natural system context. Falkenmark and Rockström, for instance, in their paper of 2006, highlighted the urgent need for a shift in thinking in water resources governance and management. In their view, this shift is a move from a *blue* runoff focus to a broadened *green-blue* soil moisture and runoff focus (Falkenmark and Rockström, 2004). Its fundamental implication is to shift to seeing precipitation rather than runoff as the primary water source. Moreover, it implies a shift from applying IWRM solely as a runoff-based management framework to implementing it as a rain-based water management framework. Furthermore, Falkenmark and Rockström

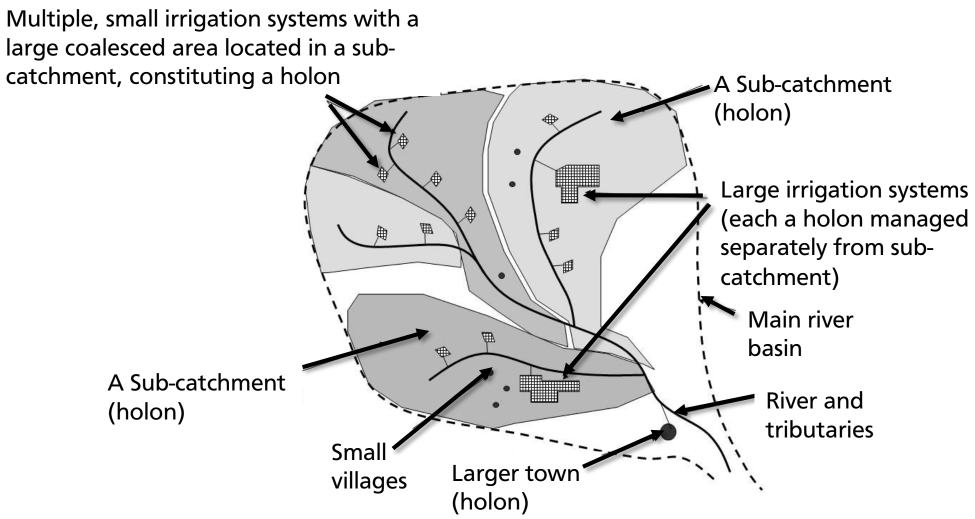

Figure 2.4: Schematic illustration of nested holons within a river basin (Lankford and Hepworth, 2010)

(2010) argue that in contrast to a conventional blue water management approach where the focus is placed at the river basin level, the application of an integrated *green-blue* approach also implies a different scale of focus. Representing a downward extension of the basin focus, the *green-blue* approach puts a "stronger emphasis on the smaller catchment in which soil moisture operates and generates a multitude of ecosystem services for humans and nature" (Falkenmark and Rockström, 2010). According to Falkenmark and Rockström (2010), actions at this meso-scale level (above the farmer's fields but below the river basin) "creates choices for partitioning rainfall and allows consumptive use of green water to be properly brought into the water balance and to influence management interventions". Moreover, this intermediate level allows "a more explicit analysis of conflicts of interest and trade-offs between green and blue water use, and between upstream and downstream users and uses" (Falkenmark and Rockström, 2010).

This down-scaling of water management from basin to catchments and sub-catchments incorporates management actions with regard to the partitioning points of green and blue water pathways. Furthermore, policies have to be developed that are well adapted to the management of rain as the primary water source (see Table 2.5). Consequently, Falkenmark and Rockström (2010) stress that two fundamental shifts are necessary for water planners: "(1) focus on precipitation as the planning resource and (2) bottom-up application of planning, starting from the local sub-catchment, i.e., the small, often ephemeral tributaries forming social-ecological units ranging in size from hundreds of hectares to hundreds of square kilometers, and moving on to catchments".

At this point there are clear parallels between the approach of Pahl-Wostl et al. and the one of Falkenmark and Rockström in using SES as its theoretical background. While Pahl-Wostl et al. derives implications for how to address the human system, Falkenmark and Rockström deduce implications on how to deal with natural system's characteristics.

According to Falkenmark and Rockström (2006), both the focus of IWRM and the scale at which it is implemented have to be redefined. A new focus has to consider the *full* water balance of blue and green water as *manageable*. In places where rain-fed agriculture is of significant importance, as in developing countries, "the scale of focus should more prominently be on the smaller catchment or watershed scale, which corresponds better to the scale relevant to the farmer" (Falkenmark and Rockström, 2006). Thus, the incorporation of land use decisions as important water decisions into the concept of IWRM is essential.

Everard and Powell (2002), in a similar manner to Falkenmark and Rockström (2006), claim that river catchments should be regarded as *living systems* where water and land are interdependently connected. Thus, management of river catchments requires a move from a reactive to a systemic approach to the water environment. For Everard and Powell (2002) the "acknowledgment of the central importance of ecosystem functions, the protection of these functions through management action, valuing them appropriately, and taking longer-term and wider-scale perspectives in management decisions" are essential for a systemic approach. In order to move in thinking and action there are seven shifts that society needs to undertake to move from a reactive to a systemic approach to the water environment:

- From anthropocentric to ecocentric - The functioning of the ecosystem, and not merely the human use of it, needs to be central to our thinking.
- From downstream 'fixes' to systemic understanding of catchments. The benefits to society provided by intact ecosystems must be quantified at the catchment scale, and properly valued by development decisions.

	New	Conventional
Planning	Sources = rainfall forming green water in soil and blue water in rivers/aquifers	Sources = water in rivers and aquifer
	Planning of management options as a continuum from rain-fed to irrigated agriculture	Planning focus on water allocations for irrigation, industry, and domestic water supply
	Cross-scale focus - down-scaling from river basin to catchment and local level	River basin plans
	Integrated land and water resource management (ILWRM)	IWRM
Policies	New environmental policies, including green water needs to sustain terrestrial ecosystems	Environmental impact assessment
	Green-blue trade-offs between up- and downstream (e.g. Water Act, Republic of South Africa)	Water in aquatic ecosystems, environmental water flow
	Forest plantation fees	Demand management
	Green water credits (GWC)[1]	Water laws regulating blue water distribution

[1] The GWC concept builds on the growing evidence that improved land and water management upstream cannot only improve green water benefits there (e.g., through increased farm yields) but also release more useful and better quality blue water to downstream uses, e.g. base flow replacing storm flow, less sediment (Dent and Kauffman, 2007).

Table 2.5: Conceptual differences between the past blue water-based management of water resources and the new green-blue water-based management of land and water resources (Falkenmark and Rockström, 2010)

- From evaluating single-function benefits to accounting for multiple benefits. Benefits stemming from the water environment, including amenity, water retention, storage and groundwater exchange, chemical purification, reduced greenhouse gas emissions, better flood storage and attenuation, fish recruitment, and nature conservation need to be factored into decision-making.
- Focus on long-term rather than short-term implications.
- From management solutions applied in isolation to ’building blocks’ contributing to catchment functioning. These potential ‘building blocks’ include sustainable drainage systems, buffer zones, re-meandering of rivers, or full-scale river restoration.
- From maintenance and mitigation to restoration.
- From consultation to consensus-building” (Everard and Powell, 2002).

The ecosystem-based approach proposed by Everard and Powell (2002) calls for management actions at the local level which “also contribute to the protection or restoration of natural functions at catchment scale” (Everard and Powell, 2002). This idea of interdependent sub-units that ought to be managed from bottom-up is similar to the proposal of Lankford and Hepworth’ *holons* as nested sub-systems for management.

2.5 Summary

Since its emergence at the beginning of the 1990s, the concept of IWRM has increasingly gained acceptance as a normative framework to guide the way how water and related land resources have to be managed. A large number of countries has now embarked on an implementation process of IWRM through the passing of IWRM conform water legislation and respective policies, creating an enabling environment. At national levels new institutional roles have been defined and specific organizations have been created to improve cross-sectoral cooperation. However, as global assessments have repeatedly confirmed, this process has taken a long time and IWRM implementation, at this stage, has largely remained without taking the subsequent step of operationalization in the practical application of participatory, stakeholder involving management instruments. Especially in developing countries, standard ‘packages’ of IWRM implementation do not take the prevailing resource constraints in these countries sufficiently into account.

The struggles towards IWRM implementation have been critically observed by a broad scientific community. While the scientific discourse, at first, focused on the complexity of the concept itself, it has subsequently addressed the problematic implications it has in practice if the concept is ‘fully’ implemented in an all-integrating, comprehensive manner. The recognition that specific contexts have to be considered led to more pragmatic and problem-oriented approaches towards IWRM implementation, known as *light*, *expedient* or *adaptive* IWRM. In awareness of the need for a more *adaptive* approach, local contexts of action are seen as fundamental to achieving effective change in water resources management. To put IWRM into context, this means moving from global, exogenous ‘solutions’ to local, endogenous plans of action (Beveridge et al., 2012). Thus, the key for improving IWRM implementation is to address the institutional and political challenges typically encountered when implementing integrated approaches: e.g. problems of institutional interplay and spatial fit, lack of participation, equity and accountability, as well as the general mismatch with needs and conditions in specific places (Beveridge and Monsees, 2012). Therefore, a problem- and people-orientated approach is needed that avoids the pitfalls of the 'one-size-fits-all' attempts which often characterize mainstream implementation.

Both Pahl-Wostl et al. and Falkenmark et al. highlight the importance of social-ecological-system's interactions in order to improve IWRM implementation from bottom-up. Both scientific communities acknowledge that a context-specific and problem-oriented approach is necessary to achieve this. Although both recognize the interconnectedness of human and natural systems Pahl-Wostl et al. put a stronger focus on improvement in the integration within the human system while Falkenmark et al. do this with regard to the natural system. Humans are the central drivers of ecosystem change but at the same time firmly dependent on the goods and services that ecosystems provide. This interdependency has to be reflected in IWRM as well. Within river basins this interdependency has to be recognized and addressed, thus, the water governance challenge of IWRM as IRBM has to take due care when linking human society and natural ecosystems.

3 Methodological conceptualization of Integrated Water Resources Management implementation

According to the implementation assessments and the broad scientific discourse, IWRM implementation is essentially a question of governance and management at the right scale. Moreover, since decentralization and the river basin approach are imperatives in IWRM implementation, it is more precise to refer to a multi-level governance[1] challenge. Therefore, Mitchell (1990), among others, argues that the IWRM concept may be applied at normative, strategic and operational levels. According to Mitchell, there has been much progress, although sometimes mostly rhetorically, at the normative level by defining what ought to be done to achieve IWRM. Examples are the numerous national IWRM plans, strategies and also general laws on IWRM at the national level. The strategic level subsequently represents the principal level of governance in practice and is concerned about what can be done to actually implement IWRM, e.g. implementation in a normative sense as *full* IWRM or *light* IWRM as context-specific interpretation of IWRM. Finally, the operational level is concerned with the operationalization of the targets defined at the governance, i.e. strategic, level. According to Mitchell (1990), what actually will be done is defined at this level. The IWRM process should take place at all of these levels and it is important to be aware that the IWRM process often moves from one level to the next. Scheuchzer et al. (2012) identify two different levels of IWRM as important for implementation: Integrated Water Governance at the national, i.e. strategic level, and Integrated River Basin Management, i.e. operationalization of IWRM at the river basin level.

In the context of IWRM implementation, achievements at the strategic and operational level (cf., Grambow, 2013, also normative and operational management) - the river basin level and below - have been especially difficult. This difficulty has its origin in the significant changes in the governance system of natural resources implied by the IWRM process (Rogers and Hall, 2003; Dombrowsky, 2005). Changes in the governance system in turn require changes in management instruments as well, i.e. how water governance is implemented, because management instruments are the means to implement changes in the governance system. IWRM challenges traditional governance systems, which implies the use of different management instruments to meet the targets of IWRM governance. These challenges, in this context, refer to integration as cooperatively balanced interests among all relevant stakeholders (i) in a cross-sectoral manner (comprising different water sectors, including use and conservation of the resource), (ii) in a spatial manner (within a relevant area within a river basin across administrative boundaries), and (iii) in an institutional manner (across all relevant governmental levels) (Huppert, 2005).

Thus, the general subject of this chapter is the methodological conceptualization of IWRM implementation problems in practice. Based on this conceptualization of implementation gaps, the requirements for specific instruments to achieve further implementation are assessed in order to develop a solution statement. For a better understanding of the discussion in this chapter it is useful to define several terms with regard to governance analysis. A useful definition of the term governance, as it is used in this dissertation, is provided by Hufty (2011):

> *"Governance refers to a category of social facts, namely the processes of interaction and decision-making among the actors involved in a collective problem that lead to the creation, reinforcement, or reproduction of social norms and institutions".*

Hence, governance is about how each society develops its own ways of making decisions and resolving or avoiding conflicts. Elements of governance such as decision-making processes, social norms, and institutions allow members of a society to live together and cooperate, even without an omni-present state (Hufty, 2011). Thus, contrary to the concept of *political systems* and the traditional idea of *politics*, governance does neither presuppose vertical authority nor regulatory power. Interpreting governance in this way, it refers equally to formal and informal, vertical and horizontal processes, without predefined preference. According to Hufty (2011) using this governance perspective permits the inclusion of a large variety of social processes. This interpretation is especially useful in the context of IWRM implementation in developing countries where governments and formal rules are often weak in performance.

Consequently, water governance can be understood as a set of systems that controls decision-making with regard to water resources development and management. Water governance, thus, is more about the way how, i.e. by whom, and under what circumstances decisions are made than about the decisions themselves (Moench et al., 2003). According to the OECD (2011b), water governance "covers the manner in which roles and responsibilities (design, regulation and implementation) are exercised in the management of water and broadly encompasses the formal and informal institutions by which authority is exercised". Governance understood as composed of a combination of formal and informal institutions is referred to as *distributed* governance (cf., Kooiman, 1993).

[1] The OECD defines multi-level governance as the explicit or implicit sharing of policy-making authority, responsibility, development and implementation at different administrative and territorial levels, i.e.: i) across different ministries and/or public agencies at central government level (upper horizontally); ii) between different layers of government at local, regional, provincial/state, national and supranational levels (vertically); and iii) across different actors at the sub-national level (lower horizontally) (OECD, 2011b).

In the context of this dissertation, the term *institution* is defined as sets of rights, rules, and decision-making procedures that define social practices, assign roles to the actors in these practices, and guide interactions among the occupants of individual roles (Young, 1999, 2002b). More broadly said, an institution may be represented in any structure or mechanism of social order and cooperation governing the behavior of a set of individuals within a given community. Moreover, institutions are constructed with a social purpose (produced by collective human choice, though not directly by individual intention), transcend individuals and intentions by mediating the rules that govern cooperative living behavior. Typical examples for institutions in this sense are structures of property rights[2], electoral systems, and practices relating to resources use and conservation. Institutions form, thus, a fundamental part of environmental management (Young, 1999). Additionally, institutions are the central subject of New Institutional Economics, a branch of economic theory that attempts to extend economics by focusing on the social and legal norms and rules, hence, institutions that underlie economic activity. Comparative institutional (economical) analysis is applied to make recommendations about efficient internalization of externalities and institutional design. Among other, important elements of this analysis are property rights, transaction costs, modes of governance, social norms, ideological values, and enforcement mechanisms.

In contrast to institutions, organizations are construed as material entities with budgets, offices, employees and usually legal personality. Thus, according to Young (2002b), organizations may represent actors that typically emerge as stakeholders whose activities are guided by the rules of the game defined by institutions in which they participate. Examples of organizations are the World Bank, the Global Water Partnership or even local municipalities and water user organizations. Furthermore, Young stresses that "institutions can and do vary widely in terms of a range of dimensions, including functional scope, spatial domain, degree of formalization, stage of development, and interactions with other institutions". An environmental or resource regime (not to be confused with an authoritarian government or dictatorship) consists of a set of institutions that deal explicitly with environmental or resource issues. Amezaga (2006) stresses that organizations can be conceptualized as institutions or as "institutionalized" organizations. In the context of water institutions and organizations, Amezaga refers to RBOs, water companies, or ministerial departments as "institutionalized" organizations. Moreover, the author summarizes water institutions as a combination of: policies and objectives, laws, rules, and regulations, organizations, their bylaws and core values, operational plans and procedures, incentive mechanisms, accountability mechanisms, and norms, traditions, practices and customs (Amezaga, 2006). With regard to the core meaning of the term "institutions" and the emphasis in the popular usage of the term associated with "organizations", Bandaragoda (2000) proposes a broad interpretation of the term in the context of IRBM. In his interpretation the institutional framework for water resources consists of established rules, norms, practices, and organizations that provide a structure to human actions related to water management (Bandaragoda, 2000). Amezaga (2006) highlights that in this interpretation established organizations are to be considered as a subset of institutions.

As Young (2002b) highlights, it is important to distinguish between thin perspectives and thick perspectives on institutions. A thin perspective on institutions is limited to those institutions that are articulated in constitutive documents, e.g. constitutions, treaties and contracts (Young, 2002b). This perspective objectively describes the features of institutions as rules of the game in form of *rules on paper* (North, 1990). A thick perspective on institutions additionally includes "common discourses in terms of which to address the issues at stake, informal understandings regarding appropriate behavior on the part of participants, and routine activities that grow up in conjunction with efforts to implement the rules" (Young, 2002b). This thick perspective is important to take into account since social practices "ordinarily evolve over time in ways that are not easy to trace to their constitutive foundations" (Young, 2002b). Thus, a thick perspective is more about *rules in use* which are only partly reflected in *rules on paper*. Especially in the context of developing countries and in the view of the importance of informal economies and rules there, it is important to acknowledge the presence of all rules in use. The noncompliance of regulatory laws and other command and control measures, typical problems of governmental regulation in many developing countries, is another important argument to take a thick perspective. An important point of difference between the thin and thick perspective is that the thick definition treats behavioral consequences as a defining characteristic of institutions. Therefore, it omits rules on paper that do not influence social practices and adds *de facto* practices that do not rest on formal constitutive agreements (Young, 2002b). Thereby, "the thin definition directs attention to matters of compliance or conformance, whereas the thick definition focuses on a broader range of behavioral patterns arising in conjunction with the operation of social practices" (Young, 2002b).

Since institutions pursue a social purpose, so do environmental regimes. In the context of this thesis, the social purpose of the considered environmental regime is an integrated management of water resources as a means to ensure their sustainable use.

Although the terms water governance and water management are sometimes used interchangeably, it is useful to distinguish them. Water governance is concerned with the system and its components which are to be governed with

[2] Property rights are a theoretical construct in economics to determine how a resource is used, and who owns that resource - government, collective bodies, or individuals (Alchian, 2008). In the context of regulating the use of the environment implicit or explicit property rights can be created either through prescriptive command and control approaches (e.g. limits on input/output/discharge quantities, specified processes/equipment, audits) or by more flexible market-based instruments (e.g. taxes, transferable permits or quotas) Guerin (2003).

a core focus on institutions as well as (institutionalized) organizational structures and their efficiency. Thus, water governance determines what is being governed placing targets from a systems point of view. In order to express targets, water governance requires the design of public policies and institutional frameworks that are socially accepted and able to mobilize (social) resources in support of them (Rogers and Hall, 2003). As water governance involves the balancing of various interests facing political realities, it is clear that there is a profoundly political element to governance (cf., Allan, 2003a).

However, the term water management refers to the operational activities for meeting specific targets of water governance. In the context of IWRM, management is here basically about operationalization of IWRM in practical terms, e.g. through application of a river basin approach, cross-sectoral collaboration, stakeholder involvement and participation. Hence, water management is about how, i.e. through what kind of instruments, water governance in terms of institutional / organizational goals and objectives are attained in an effective and efficient manner through planning, organizing, leading and controlling the institutional / organizational resources (cf., Malano and Van Hofwegen, 1999).

Implications for general institutional design and appropriate operationalization instruments for implementation arise from the findings of international IWRM implementation assessments as well as from scientific discourse (see Sections 2.1 - 2.4 respectively) and can be summarized as follows:

- Success of *standard packages* for IWRM implementation has remained limited to the creation of national enabling environments and formally assigned institutional roles. Several implementation gaps (see Table 3.1 for a useful categorization of different implementation gaps provided by the OECD) persist in terms of implementation of IWRM at the strategic and operational level.
- It has now been widely acknowledged that it will probably not be possible to successfully achieve IWRM operationalization blue-prints of full integration without relation to specific implementation contexts. Hence, top-down approaches and command and control instruments are generally discarded in this context.
- A more adaptive approach towards IWRM is necessary in order to cope with prevailing complexity and persisting fragmentation in water resources management. This adaptive approach requires a shift in emphasis towards "the management of processes and people" (UN-Water, 2012a) based on continuous learning about the natural system to be managed and the human system's response to it in order to reduce uncertainty and risk. Furthermore, "better use should be made of the natural environment as a component of water resources infrastructure" (UN-Water, 2012a), e.g. "the role of land cover (vegetation) and soil in reducing hydrological risk illustrates the need to rethink water storage in ecosystem terms" (UN-Water, 2012a).
- This adaptive IWRM implies governance and management based on institutional reform, incentives and behavioral change requiring multi-disciplinary and multi-sector collaboration.

According to these findings, the focus of further IWRM implementation efforts has to be placed on improvements in water governance through effective (formal and informal) institutions as well as on appropriate management instruments to operationalize them. On the one hand, institutions have to respond to specific contexts (natural and human system properties and especially their interactions) and, on the other hand, a gradual shift towards policies based on institutional reform, incentives and behavioral change through appropriate management instruments, sensitive to prevailing constraints, is necessary. Hence, instruments to overcome the implementation gaps (operationalization) should be based on natural and human system's contexts, thus, considering people and ecosystems (participatory and ecosystem-based) - representing the relevant governance system.

The OECD developed a multi-level approach for governance of public policies in decentralized contexts "to look at ways to improve capacity and co-ordination among public stakeholders at different levels of government to increase efficiency, equity and sustainability of public spending" (Charbit, 2011). This methodology has been tested in the context of water policy implementation of OECD and non-OECD countries in order to diagnose multi-level governance challenges and examples of tools used by countries to bridge co-ordination and capacity *gaps* (OECD, 2011b; Akhmouch, 2012). Table 3.1 illustrates the identified IWRM implementation *gaps* and respective implications to overcome them. Although described in more general terms, these *gaps* reflect the principal IWRM implementation obstacles identified in Sections 2.1 and 2.2.

This chapter addresses the IWRM implementation gaps identified in Section 2.4 with respect to challenges to the water governance system on the one hand and concerning requirements for management instruments to operationalize IWRM on the other hand. The IWRM governance challenge will be analyzed based on an analytical framework for environmental governance presented by Young (1999). In an additional step, and as a response to the governance challenges, requirements for management instruments based on an actor's perspective and incentives for behavioral change of involved actors are discussed.

The concept of Young is used because, contrary to other analytical frameworks for environmental institutions, the characteristics of the environmental resource, i.e. the ecosystem, to be managed are an intrinsic design principles of Young's concept. Other analytical frameworks for institutions of environmental management generally focus more on socio-political factors and treat the characteristics of the environmental resource as contextual factors and thus are not

Governance gap	Description	Implications
Administrative gap	Geographical "mismatch" between hydrological and administrative boundaries	Need for instruments to reach effective size and appropriate scale
Information gap	Asymmetries of information (quantity, quality, type) between different stakeholders involved in water policy, either voluntary or not	Need for instruments for revealing and sharing information. Sectoral fragmentation of water-related tasks across public authorities
Policy gap	Sectoral fragmentation of water-related tasks across ministries and agencies	Need for mechanisms to create multidimensional/systemic approaches, and to exercise political leadership and commitment
Capacity gap	Insufficient scientific, technical, infrastructural capacity of local actors to design and implement water policies and relevant strategies	Need for instruments to build local capacity
Funding gap	Unstable or insufficient revenues undermining effective implementation of water responsibilities at sub-national level, cross-sectoral policies, and investments requested	Need for shared financing mechanisms
Objective gap	Different rationales creating obstacles for adopting convergent targets, especially in case of motivational gap (lack of political will to engage)	Need for instruments to align objectives
Accountability gap	Difficulty ensuring the transparency of practices across the different constituencies, mainly due to insufficient users' commitment, lack of concern, awareness and participation	Need for institutional quality instruments, instruments to strengthen the integrity framework at the local level and instruments to enhance citizen involvement

Table 3.1: Key implementation gaps in water policy; based on Charbit (2011) and further elaboration by OECD (2011b)

reflected explicitly in design principles (cf., Edwards and Steins, 1999). This structural assessment is carried out in order to conceptualize the requirements for necessary institutions (i.e. management instruments) to overcome the IWRM implementation gaps. It abstractly describes the general problem setting, why IWRM implementation at the river basin level and lower is challenging and why implementation problems occur, with the aim to shed more light on implementation gaps at the governance level in a structured manner. The aim is to deploy a general analytical framework, based on prior findings of Young and others to describe the problems of integration, cooperation / collaboration and finding the right scale of both in a context-specific manner. In doing so, the requirements to cope with generic IWRM implementation problems are provided without addressing particular contexts nor specific problem-settings. This is supposed to provide guidance to identify appropriate management instruments addressing these generic problems by being flexible in addressing different contexts and problem-settings.

Following the IWRM trajectories discussed in Section 2.4, it is assumed that IWRM implementation is a matter of governance and management of complex Social-Ecological System. Both, the adaptive management approaches with a stronger focus on social and institutional issues (Timmerman et al., 2008; Pollard and Du Toit, 2008; Pahl-Wostl, 2009; Lankford and Hepworth, 2010; Pahl-Wostl and Kranz, 2010; Engle et al., 2011; Pahl-Wostl et al., 2012) and those with a stronger focus on ecological issues (Everard and Powell, 2002; Pinkham, 2002; Falkenmark and Rockström, 2004, 2006; Smith and Barchiesi, 2008; Falkenmark and Rockström, 2010) refer to SES and highlight the importance of context-specific approaches, thus representing *expedient*, *light*, *adaptive* approaches of IWRM, based on specific characteristics of the natural and human system as well as, more importantly, their interactions, to achieve better IWRM implementation.

The analytical framework of fit and interplay by Young (1999) is consistent with the SES theory, since both integrate a systems approach and adaptive management, emphasis on linkages and feedback (Holling, 1978), and improvement of performance of natural resource systems based on an institutional and property rights approach, a people-orientated approach which focuses on the resource user rather than on the resource itself (Berkes et al., 2000). Table 3.2 describes Young's framework of institutional fit and interplay in general as well as in the context of IWRM implementation and relates it to the OECD implementation gaps (see Table 3.1).

Accordingly, the approach in this context integrates two streams of resource management thought (Berkes et al., 2000). One is the use of a systems approach and adaptive management with emphasis on linkages and feedback controls (Holling, 1978). The system approach recognizes that natural resources cannot be treated as discrete entities in isolation from the rest of the ecosystem nor the social system. Thus, social and ecological systems are understood as truly interdependent and

Conceptualization by Young	Description	IWRM context	OECD multi-level governance gaps
Problem of fit	The effectiveness of social institutions is a function of the match between the characteristics of the institutions themselves and the characteristics of the biogeophysical systems with which they interact	Problems of incompatibility between institutional arrangements (e.g. municipal land use planning, use of water resources, up- and downstream communities) and biogeophysical systems	Administrative gap
Problem of interplay	The effectiveness of specific institutions often depends not only on their own features but also on their interactions with other institutions either at the same level (horizontal interplay) or across levels of social organization (vertical interplay)	Problems of interaction of different institutions (e.g. different *rules* for water users: agriculture, water supply, power generation, sanitation), cooperation and collaboration among stakeholders	Policy gap, Objective gap, Accountability gap

Table 3.2: The framework proposed by Young (1999) to analyze institutional challenges of environmental governance and related OECD governance gaps

constantly co-evolving. Several scientific disciplines, i.e. human or social ecology and ecological economics, are centered on this approach.

The other stream of thought deals with improving the performance of natural resource systems. It places emphasis on institutions and property rights representing a people-oriented approach which focuses on the resource user rather than on the resource itself - *resource management is people management* (cf., Berkes et al., 2000). Concepts incorporated in this approach relate to institutions and collective action, community-based resource management and institutional learning and resource management (Berkes et al., 2000).

This analytical framework refers to the identified *administrative gaps* in water policy as problems of fit (Section 3.1) and to policy, objective and accountability gaps as problems of interplay (Section 3.2). Although information, capacity and funding gaps are also addressed in the context of problems of fit and interplay, these gaps will be assessed further in terms of principal problems of operationalization in the context of IWRM implementation in developing countries (Section 3.3). This is done to highlight the special circumstances in these countries (e.g. of weak state, problems of corruption and public finance) that present additional obstacles towards IWRM implementation.

This framework is suitable to analyze the implementation obstacles that arise while implementing IWRM at the river basin level and below, because it directly addresses the interdependence of the natural and the human systems as social-ecological-systems. Accordingly, problems of fit address the principal challenge of institutions to match the ecosystems and their properties which they manage. This can be interpreted as an integration challenge within the natural system (cf., GWP, 2000) focused on by Falkenmark et al. The institutional challenges that result from improving fit are addressed as problems of interplay between different institutions. This can, in turn, be interpreted as integration challenge within the human system (cf., GWP, 2000), the focus of Pahl-Wostl et al., Lankford et al. and others. The concept of institutional fit and interplay, thus, is applied as an analytical framework in response to the need for adaptive, context- and site-specific solutions that are required to overcome IWRM implementation gaps.

The problems of fit and interplay are interdependent, as will be explained in the following sections, thus, allowing to identify instrument requirements that explicitly address the challenge of interdependence of natural and human systems. Hence, this will be the first, context-specific step of identification of institutional design requirements based on an SES-view which will be followed by an operationalization analysis based on respective actor / stakeholder views. This second analytical step focuses on incentive measures to change actor's behavior. An actor orientation to institutional design is essential since participation and stakeholder involvement can only be achieved if actors can be actively engaged.

Figure 3.1 illustrates the methodological conceptualization of the IWRM implementation (illustrated in the upper governance sphere of the Figure) and operationalization (illustrated in the lower management sphere of the Figure) based on the problem analysis of Chapter 2. Hence, this chapter conceptualizes the analyzed governance and management challenges along IWRM implementation. The principal objective is to:

- Address the governance challenges conceptualized as interdependent problems of institutional fit and interplay by finding appropriate governance scales; and to
- Address the operationalization challenges conceptualized as operational constraints and appropriate incentives by finding appropriate management instruments

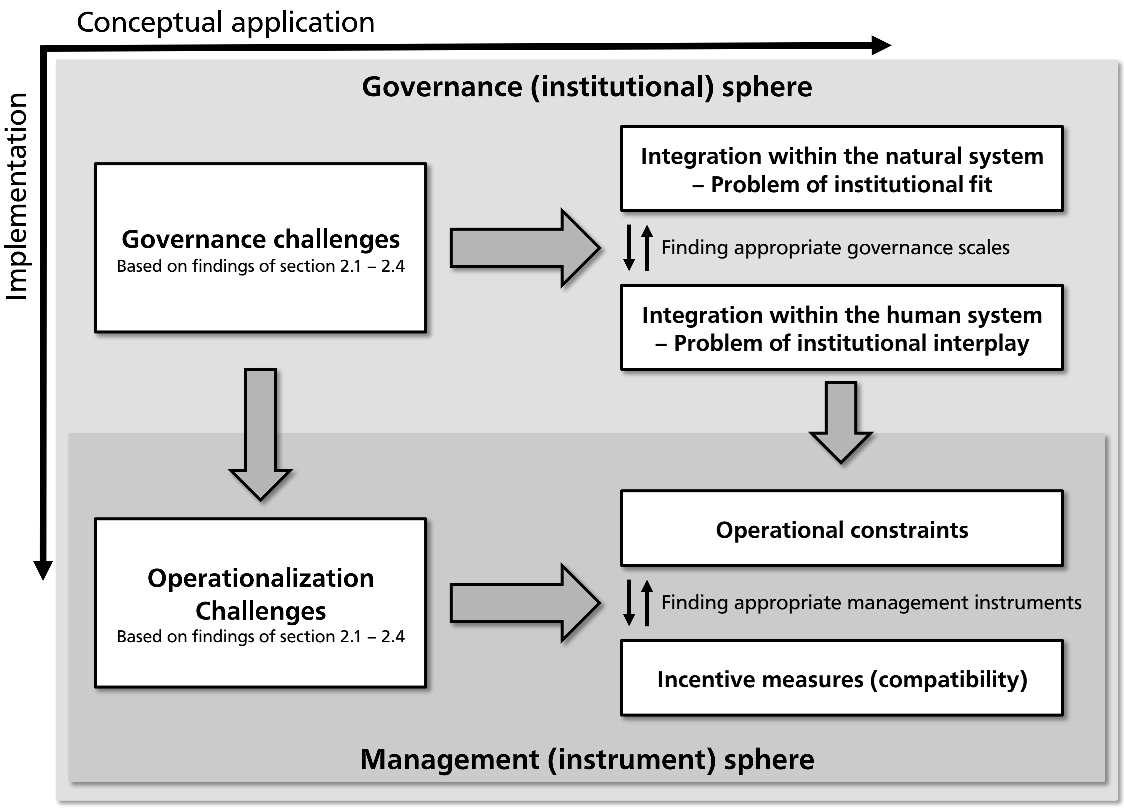

Figure 3.1: Overview of IWRM implementation assessment (Source: Author's work)

3.1 Problems of institutional fit

The notion that there is no 'one-size-fits-all' when it comes to designing environmental regimes to solve a variety of environmental problems has been expressed prominently by Young (2002b). Hence, the problem of fit is generic, it occurs whenever and wherever humans interact with bio-geophysical systems, regardless of the scale (e.g. large river basin or small tributary watershed or even smaller). Young (2002b) stresses the importance of fit of environmental regimes to ecosystem dynamics and properties for their sustainable management. It is important to look beyond the immediate and stationary problems of overuse or point-source pollution and instead take a systems approach in responding to wider causal effects and identifying linkages between these and characteristics of human systems (Moss, 2006). Such a system's approach considers also linkages within the natural and human systems, e.g. diffuse pollution, groundwater recharge, flow regimes etc. To illustrate this, Young (2002b) gives an example of the calculation of sustainable yields from stocks of fish. These calculations have to consider the interactions (e.g. predator-prey-relationships) with other ecosystem components (other species or physical; e.g. temperature; and biological conditions; e.g. reproduction rates) to be sustainable. This is an argument in favor of holistic ecosystem management. If regimes do not consider the interactions and the chaotic behavior of the ecosystems they govern, Young (2002b) argues that "the predictable result is a problem of fit that leads to unintended consequences that threaten or even preclude sustainability" (Wilson et al., 1994). Thus, problems of fit "concern the failure of an institution or a set of institutions to take adequately into account the nature, functionality, and dynamics of the specific ecosystem it influences" (Ekstrom and Young, 2009). Accordingly, Young (2008) stresses that "the effectiveness of social institutions is a function of the match between the characteristics of the institutions themselves and the characteristics of the bio-geophysical systems with which they interact".

Solutions to the problem of fit strongly influence the performance of institutions that govern human and environment relations (Young, 1999, 2008). Moreover, Brown (2003) argues that the problem of fit is of particular importance in the context of reconciling environmental conservation and development goals. The basic assumption is that the effectiveness of an institution is diminished where its characteristics do not match the characteristics of the biophysical systems it addresses, e.g. the degree to which an institution's area of operation covers the same geographical area as the natural resource it is designed to influence (Moss, 2001). The challenge of environmental governance in particular, is that the spatial context of natural resources cannot easily be altered. Therefore, institutional arrangements need to adapt to the properties of the specific resource to be managed (Moss, 2006). This kind of boundary problems between natural and human systems, described here as a problem of fit, has received much attention in the literature on the management of common pool resources (Ostrom, 1990; Ostrom et al., 1994). Thus, attempts to redesign institutions to follow the characteristics of specific ecosystems are not just an obstacle to effective governance but the heart of the problem (Moss, 2006; OECD, 2011b).

According to Young (2002b) an assessment of the principal properties of the relevant ecosystems should be the starting point to solve problems of fit. Hence, Young (2002b) identifies a number of important ecosystem properties and groups a range of relevant ecosystem properties for problems of fit under three broad headings: ecosystem structures, processes, and linkages (Table 3.3).

Property category	Ecosystem property	Description
Structures	Complexity	Elements playing functionally distinct roles that are essential to their maintenance and resilience
Properties of elements of ecosystems and their relationships	Homogeneity	Degree of similarity among individual elements
	Interdependence	Tightness of links or couplings among elements or subsystems
Processes	Productivity	Metabolic processes with varying length, magnitude, and regularity
Ecosystem dynamics that develop, maintain, or transform individual systems through time	Growth	Developmental process
	Stabilization	Capacity of an ecosystem to recover from disturbances or to return to some earlier state
	Change	Change may be cyclical, episodic, or chaotic. Both, the nature of triggers and the extent to which resultant processes are path dependent or constrained by events occurring within a system during previous times vary greatly
Linkages	Boundary conditions	Permeability of boundaries
Connections between ecosystems	Transboundary interactions	Interaction with neighboring and other ecosystems
		Exogenous impacts

Table 3.3: Ecosystem properties relevant to the problem of fit (Young, 2002b)

This starting point highlights the importance of measuring and monitoring ecosystems the properties of ecosystems or to gain knowledge about them from other sources that may be based on experiences of local experts in order to solve problems of fit. It has, thus, much in common with the social learning concept of adaptive management discussed in Section 2.4.

In a next step, institutional arrangements that fit the contours and inherent properties of bio-geophysical problems should be designed and built. Young (2002b) highlights that “this is easier said than done”. For ecosystems, understood as complex systems where everything is related to everything else, “a specification of boundaries separating distinct ecosystems is ultimately somewhat arbitrary and may emerge as a barrier to addressing important issues” (Young, 2002b).

Furthermore, Young (2002b) stresses to be “wary of an argument that treats the properties of ecosystems as objective realities or facts of life that are immune from the effects of social construction”. Accordingly, he argues that “it seems clear that regimes governing human harvesting of renewable resources that do not take into account variations among ecosystems in these terms are asking for trouble by ignoring the problem of fit; that is, by failing to devise institutional arrangements that are constructed in such a way that they are compatible with key properties of relevant ecosystems”.

Galaz et al. (2008) develop these ecosystem properties in the context of problems of fit further and derives different types of misfits. These different types of institutional misfits, concerning spatial and temporal fit, threshold behavior and cascading effects of ecosystems, are described in Table 3.4 (categories *threshold behavior* an *cascading effects* in grey color are not further considered in the context of this dissertation).

According to this categorization, the problem of fit articulates how well environmental institutions match spatial or temporal scales of ecosystems and account for functional ecosystem processes (Galaz et al., 2008). Misfits of environmental regimes and ecosystem properties have contributed substantially to the deterioration of ecosystem services (Young, 2002b,a; Ekstrom and Young, 2009). According to Galaz et al. (2008) the problem of fit concerns the failure of an institution or a set of institutions (regime) to take adequately into account the nature, functionality, and dynamics of the specific ecosystem it influences. Governance *gaps*, i.e. more precisely administrative gaps (see Table 3.2) in the context of problems of fit, prevail where institutional mechanisms that account for links within and among resource use sectors and significant properties of the ecosystem are lacking (Galaz et al., 2008; Ekstrom and Young, 2009). Misfits are present where institutions leave these *gaps*, for instance, by not completely covering the ecosystem that comprehends the resource or the resource use they are designed to manage by institutions (Hoel et al., 2005; Ekstrom and Young, 2009). Besides the institutional matching of the ecosystems properties, thus, it is also important to take the resource use and the alternation of ecosystem properties through human action into account.

Type of misfit	Definition and mechanism of misfit
Spatial extension	Institutional jurisdiction too small or too large to cover or affect the areal extent of the ecosystem(s) subject to the institution Institutional jurisdiction unable to cope with actors or drivers external or internal and important for maintaining the ecosystems or processes affected by the institution; e.g. "one size fits all" institutions are designed inappropriately to local social or ecological contexts (cf., Ostrom, 1999)
Temporal dynamics	Institution formed too early or too late to cause desired ecosystem effects Institution produces decisions that assume a shorter or longer time span than those embedded in the biophysical systems affected; and/or social response is too fast, too slow, too short, or too long compared to the time taken for biophysical processes involved
Threshold behavior	Institution does not recognize, leads to, or is unable to avoid abrupt shift(s) in biophysical systems Institution provides for inadequate response to contingencies (e.g. lack of rules for action in extreme conditions) or reduces variation in biophysical systems (e.g. by removing whole functional groups of species and/or by adding anthropogenic stress such as pollution). Institutions fail to respond adequately or at all to disturbances that could have been buffered. Leads to practically irreversible biophysical shifts (Folke et al., 2005)
Cascading effects	Institution is unable to buffer, or trigger further effects between or among biophysical and/or social and economic systems Institutional response is misdirected, nonexistent, inadequate, or wrongly timed to propagate or allow the propagation of biophysical changes that entail further causative changes along temporal and/or spatial scales

Table 3.4: Types of misfits between ecosystem dynamics and governance systems; based on Galaz et al. (2008)

In his analysis Young (2002b) finds evidence that suggests a number of distinct mechanisms that result in institutional mismatches with ecosystem properties. Often two or more of these mismatches occur at the same time, thus, the sources of mismatches are not mutually exclusive (Young, 2002b). The sources of mismatches are grouped by Young (2002b) into three main categories: imperfect knowledge, institutional constraints, and rent-seeking behavior (see Table 3.5).

Category of mismatch	Sources of mismatch
Imperfect knowledge	Ignorance because of absence of usable knowledge, underestimation of exploitation impacts or lack of awareness regarding the ecosystems in questions Faulty models or misleading discourses Disregard of endogenic role of human actions in coupled human-natural systems
Institutional constraints	Embeddedness of regimes in larger or overarching social institutions (institutional linkage) Jurisdictional boundaries Implementation of lead agencies for purposes of administering a regime in practice Path dependence - diverging change velocities of environmental or technological change and institutional change
Rent-seeking behavior	Actors try to improve their individual payoffs at the expense of social welfare In the absence of rules designed to protect the public interest, private actors often exploit natural resources ruthlessly for their own benefit (*rape, ruin, and run*) Political rent-seeking on the part of leaders able and willing to pursue their own objectives regardless of longer-term consequences for sustainability of human-environment relations

Table 3.5: Sources of mismatches between regimes and ecosystems; based on Young (2002b)

The governance challenge increases with the number and types of misfits (e.g., from local spatial misfits to cross-national cascade effect misfits) as a result of the enlargement in the number of actors, spatial scales, and interactions across systems introduced by environmental and resource regimes operating at different governance levels (Young et al., 2008).

Although there are "no simple antidotes to these forces leading to the persistence of mismatches between ecosystem properties and institutional attributes", Young (2002b) presents three possible approaches to confront them. One practical approach is to establish a monitoring system of both the status of key ecosystems and the performance of major environmental regimes. This way evidence of mismatches can be revealed through continuous, detailed, and credible feedback regarding the course of relevant human-environment relations. This approach is related to the social learning in adaptive management approaches. Building substantial flexibility into provisions of environmental regimes is another approach. This approach is also incorporated in adaptive management. Finally, Young (2002b) proposes a precautionary approach “to respond to problems arising from imperfect information and institutional constraints by erring on the side of safety; that is, by building in margins of safety to ensure that exploited components of ecosystems are not pushed beyond the limits of sustainability”.

In the context of water governance an administrative gap (see Table 3.1) described as a problem of fit, thus, corresponds to a geographical mismatch between hydrological and administrative boundaries (OECD, 2011b). In the governance of natural resources, in this case water resources, the administrative gap represents a fundamental problem of fit since the logic of service provision by a governing institution is constrained by nature. This, however, is not the case for many other governance fields, e.g. health or education. Governance fields that are independent from the natural systems can therefore be much easier administrated in line with administrative boundaries of municipalities, regions and states. Moreover, poor fit of institutions in environmental governance is not a phenomenon limited to the international level, it is also true at the national and even the local level where for historical and economic reasons jurisdictions of virtually all governments match poorly to natural system’s properties (Lipschutz, 1999).

Administrative gaps in national water governance[3] principally imply mismatches at sub-national level that often obstruct water policies and complicate the relationships between elected representatives, local authorities, water agencies, resource managers and end users (Moss, 2006; OECD, 2011b). Problems of mismatches such as lack of co-operation, participation and transparency in water management are often rooted in the historical organization of water bodies along administrative boundaries, although river basins rarely obey administrative logic (Molle et al., 2007; OECD, 2011b). Hence, the enforcement of water quality regulations and water abstraction rules, for instance, is difficult where two or more institutions regulate different sections of one river, i.e. of a river basin. Furthermore, management of water resources , in the past, has often focused on either point-source or scattered and uncoordinated solutions for water pollution (e.g. end-of-pipe solutions to waste water problems), resource use or flood protection according to local administrative priorities lacking an integrated basin perspective.

Accordingly, Voigt (1997) argues that effective water resources management, in terms of both quality and quantity related aspects, depends on a governance, i.e. management concept, which reflects the complexity of water-based ecosystems, the multiple anthropogenic uses of water and the interaction between biophysical and human systems. Thus, policies which cover only a part of the water system (i.e. the river basin), a point source of pollution for instance, “[...] without considering the broader context run the serious risk of ignoring, or even creating, negative external effects” (Moss, 2006). The management of water resources on the basis of the whole river basin (longitudinally from the source to the estuary of a river and laterally from the river stream to the drainage-divide) aims at addressing the interdependence between, in particular, upstream and downstream effects, water quality and water quantity aspects, and water and adjacent land-use (OECD, 1989). In the context of IWRM, problems of fit are most prominently addressed with regard to spatial fit. The principal policy strategy to overcome the common problem of spatial *misfit* of administrative boundaries and river basin delineations is found in Integrated River Basin Management (see Figure 3.2). It is assumed that at the river basin level, apart from spatial misfits, misfits stemming from temporal dynamics, threshold behavior and possible cascading effects (cf., Table 3.4) are captured as well.

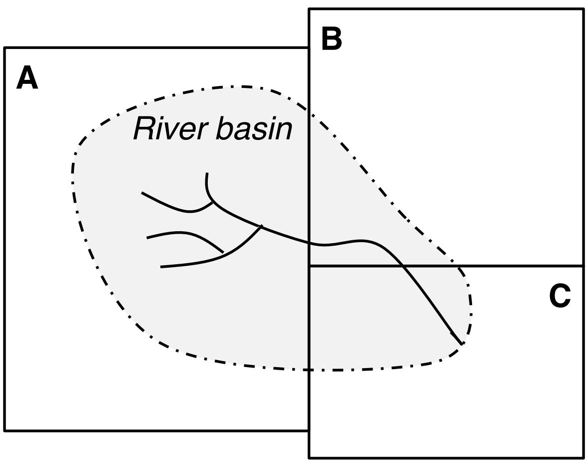

Figure 3.2: Illustration of the problem of spatial fit between a river basin and administrative boundaries of A, B and C (Source: Author’s work)

[3] Problems in the context of international river basins are not addressed in this dissertation.

Rogers (1997) argues further economically that for the internalization of externalities "[t]he river basin itself is an ideal unit of analysis to achieve this goal: it can reasonably be assumed that most externalities are captured by analysing the river basin as a single unit". Hence, the central argument for the need to solve problems of spatial fit "[...] is that lack of fit causes spatial externalities, benefiting free riders and harming others beyond the spatial reach of the responsible institution" (Moss, 2001). The concept of fiscal equivalence (Olson, 1969) addresses the problem of fit in economic terms as well, stating that those who receive a benefit from a collective good (e.g. rival but open-access water resources) need to be matched with those who pay for and decide on it. A link between the spatial extent of a problem and the spatial remit of political decision-making bodies and jurisdictions is similarly asked for in the related theory of fiscal federalism (Oates, 1999). However, the question of finance of water governance raised here has to be seen in the context of decentralization and will be addressed in section 3.3.

3.1.1 Integrated River Basin Management to solve problems of spatial fit

As highlighted in Section 2.1 and stressed in the introduction of this chapter, IWRM is a multi-level process that ranges from an international, transboundary level over national and regional levels to the local level (Lenton, 2011). While at the national level progress is made with the development of new water laws, national water policies and new institutional roles, the progress at the regional and local level is slow (see Section 2.1.2). The actual operationalization of IWRM with tangible outcomes takes place at these levels. This regional or local IWRM incorporates Integrated River Basin Management (IRBM) as a subset of the IWRM process. Hooper (2005) defines IRBM "as an integrated and coordinated approach to the planning and management of natural resources of a river basin, one that encourages stakeholders to consider a wide array of social and environmental interconnections, in a catchment / watershed context". The management of water resources according to river basins, thus, is a prominent approach to solve problems of institutional misfit. Hence, Hooper (2005) highlights the characteristics of integrated, adaptive river basin management, derived from IRBM definitions of Mitchell and Hollick (1993) and Naiman (1992):

- Coordinated activities rather than amalgamated programs of action;
- Top-down management meeting bottom-up management;
- Strategic planning rather than all-embracing efforts: being targeted and selective about actions and prioritizing work programs;
- Integrating goals rather than planning resource use and conservation by either single or multi-purpose reasons;
- Proactive rather than reactive resource use planning: looking to identify problems before they occur and being cautious in resource use;
- Using cost-effective rather than prescriptive financial management mechanisms;
- Using partnerships and cost-sharing programs wherever possible;
- Working with partners in a co-operative work environment, rather than using confrontational and directive management;
- Encouraging commitment in staff rather than using command-and-control management styles;
- Empowering local and regional decision-making rather than centralizing decisions and directing staff;
- Management based on problem-solving rather than functionality
- Having flexible organizations rather than rigid inflexible structures;
- Providing appropriate, relevant, affordable information that is relevant to IRBM;
- Using equitable management methods which are sensitive to and respect cultural needs and gender issues, and do not discriminate against catchment managers because of their distant location from the decision-making processes of other water professionals.

Hooper (2005) sees river basin decision-making as a set of principles and characteristics (see above) which need to be applied to any river basin management setting. In order to be effective the planning process, requires the following: (1) the planner has jurisdictional authority over the system being planned and (2) the planner can predict the consequences of his or her plans. Moreover, Mitchell (1990) argues that an integrated approach considering land and water resources "offers the possibility of addressing the dynamics of an ecological system, thereby ensuring that critical relationships are identified and managed". Jaspers (2003), highlights that water management not coinciding with the boundaries of the river basin is very cumbersome when conducting its principle tasks. Accordingly, it will induce authorities to monopolize the water supply sources within their area and to transfer the problem of flooding downstream. Therefore, water management on hydrological boundaries is needed because of the growing competition for water or the need to co-operate in an upstream-downstream relation for flood control or both.

However, the suitability of an entire river basin as best spatial fit is not always given. Lenton (2011) argues, for instance, that "some aspects of integration within the natural system (e.g., green and blue water, surface and groundwater management) are more relevant at the watershed and basin levels, whereas some aspects of human-system integration (such as considering water-resource policy within national economic and sectoral policies) are more relevant at the national level". Furthermore, Moss (2012) argues that on the one hand spatial misfit of institutions is often responsible for negative externalities, benefiting free riders and harming others beyond the reach of the institution responsible. On the other hand, responding to this misfit by increasing the geographical scale of institutional arrangements until all externalities are covered "often brings with it serious drawbacks in the form of unwieldy and bureaucratic structures with very little sensitivity to local or regional contexts" (Moss, 2012). Lenton (2011) too, stresses that "some problems are best addressed at the sub-basin or local level. Accordingly, the OECD emphasizes the need for instruments to reach effective size and appropriate scale as key to overcome this administrative gap (OECD, 2011b).

Consequently, although the river basin with its clearly delineated boundary for surface water appears to promise the solution to spatial misfits in water management, experiences of institutionalizing river basin management are falling short of expectations. Moss (2012) highlights three principal criticisms from literature on IRBM since the 1980s challenging the notion of creating perfect spatial fit:

- IRBM does not solve all hydrological boundary problems. The flows of surface water, not groundwater, delineate river basin boundaries. River basin boundaries are overridden where water supply networks (inter-basin transfers) or artificial waterways connect two or more river basins.
- River basin organizations lack the legitimacy and authority of democratically elected bodies of local, regional, or central government as they cover a different territory than political jurisdictions. They can experience difficulties in collaborating with policy fields/water users which are not organized around river basins, e.g. urban development, agriculture, forestry, transportation, and energy (Moss, 2004b; Mostert, 2003; Mostert et al., 2007; Pahl-Wostl, 2007).
- Water management along an ecosystem boundary has often resulted in a focus on biophysical, rather than socioeconomic, problems of water management (Mostert, 1999). A recent criticism that river basin management in practice is often neglecting participation and transparency reflects this point (Molle, 2008a; Mollinga, 2008; Saravanan et al., 2009).

Moreover, a perfect spatial fit may not be achieved since "the replacement of existing institutional arrangements by institutions oriented around biophysical systems will inevitably create new boundary problems and fresh mismatches" (Moss, 2012). Accordingly, Moss (2012) argues that "a purist pursuit of river basin management will tend to exclude from consideration factors that are not central to resolving problems of spatial fit". Again, as discussed in Section 2.4, instead of striving to achieve the ideal spatial fit across the entire river basin it is argued to consider the territorial unit of the river basin in a broader context-specific way of overlapping social, economic, political and physical spaces (Lipschutz, 1999; Moss, 2012). Accordingly, based on a comparative study of about 100 cases of river basin management worldwide the OECD highlighted that the key to solve problems of fit is to determine the rationale for the choice of spatial management unit, and to consider carefully the relative merits of adopting the river basin area as the management unit (OECD, 1989, as cited in Moss (2006)).

Nevertheless, discarding the concept of spatial fit is not an issue in the literature, instead a more pragmatic approach is prevailing. Moss (2012) states that "this [more pragmatic approach] entails accepting the existence of multiple geographies of water, with overlapping social, economic, political, cultural, and physical spaces, and the importance of collaborative and flexible ways of working across the boundaries they entail". Furthermore, Moss argues that "spatial fit, like river basin management in general, should be seen not as a panacea to environmental problems (Ostrom, 2007), but as a practice of adaptive (co-)management involving a wide range of relevant stakeholders operating in different spatial contexts and on different scales". Referring to the statements of the chapter on freshwater in Agenda 21 that states that "Integrated water resources management, including the integration of land- and water-related aspects, should be carried out at the level of the catchment basin or sub-basin" (ICWE, 1992) and the Article 26 of the Johannesburg Plan for Implementation that "the river (or water) basin should be used as the basic unit for integrating management", Lenton (2011) acknowledges that water management at the basin level has become the central focus of much of the advances in thinking about IWRM. Notwithstanding, Lenton (2011) highlights, in a similar manner as Moss (2012), also the limitations of water management at the river basin level: "while basin boundaries provide a useful way of delimiting the supply side of the equation, they are not necessarily the best means to integrate the demand side, especially since basin boundaries usually do not coincide with political or administrative boundaries. Integrating natural and human systems therefore generally requires work at other levels beyond the basin" (Lenton, 2011). Furthermore, terrestrial ecosystems that define the partitioning of rainfall within the hydrological boundaries certainly extend across these boundaries. A forest fire in a neighboring river basin, for example, will not respect hydrological boundaries, thus, impact across these poses limitations on the hydrological cycle. Nevertheless, river basins are considered as the most suitable general "management" unit because they do match significant process of surface water flows better than other management units, e.g. administrative units.

When considering sub-basins or tributary river systems, it is important to take, at least, the obvious connections within the larger river basin into account. The notion of sub-basins as nested sub-systems forming together a river basin, as proposed by Lankford and Hepworth (2010), is a useful analytical conceptualization in this context (see also Section 2.4). Uphoff (1996) claims that these *holons* - nested sub-systems - as introduced by Koestler (1967) may be considered as *whole* in themselves, and at the same time *parts* of larger wholes. Hence, certain aspects of the behavior of a *holon* may be confined entirely to that *holon*, yet systems (i.e. *holons*) at any level are constrained and controlled by the levels above and below, thus, can be considered as nested (Stephens and Hess, 1999). Considering *holons* as social-ecological systems, they represent a spatial extent of ecosystem-human system interaction. An example of a holon as a nested sub-system can be individual farmers who determine what will happen within their own farming system but must work within the social, political, technical, economic and environmental constraints imposed by a larger (e.g. village or catchment) system. Using ecosystem properties and human system interaction (use of the natural system) seems to be an interesting way to determine the appropriate management unit for the strategic and operational level. Knowledge about the natural and human system interaction is very important for this determination, thus, an adaptive and integrated approach based on social learning appears to be most adequate. Moreover, according to Cleveland et al. (1996), it is reasonable to assume for practical purpose that ecosystems function to some extend as divisible systems functioning as connected subsystems, or holons (cf., O'Neill et al., 1986). This way scientific and practical understanding of the system can be divided into tractable smaller problems. Hence, good fit between governance and biophysical systems may be based on a government structure nested across levels of administration with an adaptive capacity as suggested in research on multi-level environmental governance (Young et al., 2008). This challenge of scoping the analysis broadly enough spatially to consider all important consequences of all interactions, but narrowly enough to be efficient in making decisions about planning, operations, regulations, and other considerations represents a basic analytical challenge for IWRM. Hence, Cardwell et al. (2009) sees in the realm of water resources management the problem that plan formulation is often scoped in hydrological boundary context, "but plan evaluation of the economic and environmental consequences considers a spatial context defined by business-system interactions that may not align with the hydrology".

Since water is the management, i.e. governance subject, river basins as drainage area of rivers provide a quite a suitable spatial fit to the governance challenge. However, it is important to highlight that the river network alone only represents the *blue* water component for spatial fit, adjacent terrestrial ecosystems that determine the rainfall partitioning as *green* water flows complement the spatial fit required for good governance. Although an entire river basin may not represent the best spatial fit for all institutional challenges, the linkages of terrestrial ecosystems and river networks need to be taken into account in any case in order to achieve adequate spatial fit (cf., Falkenmark and Rockström, 2010). This is an essential requirement for institutions (i.e. management instruments) to cope with the implementation of IWRM. Since humans are an important driving force (see Vitousek et al., 1997) for terrestrial ecosystem change (green water flows) and water use (blue water flows), both natural ecosystem properties (as described in Table 3.3) and human use and alternation of them interactively define the best spatial fit. Thus, achieving good spatial fit is highly site-specific and context (i.e. use)-specific. This implication is stressed in the adaptive IWRM approaches with social and ecological focus.

Vatn and Vedeld (2012) make this point of context-specificity also in their review of Young's concept of fit and interplay when they state that "the challenge from a governance perspective is, however, not only about how to construct regimes that fit the dynamics and complexities of the resources or ecosystems. No governance structure can be evaluated without reference to an aim. Ecosystems are not fixed entities. They convey many properties and these can, at least to some extent, be re-arranged or modified to change the 'delivery' of the system. Hence, the goals we try to attain will influence what we emphasize and see as problems". Consequently, Vatn and Vedeld (2012) conclude that since "regime formulations are dominantly related to human use in some sense [...] the concept of fit needs to include references to what aspects and capacities of the biophysical system humans emphasize". In other words, the definition of spatial fit, hence the corresponding socially constructed government regime, is not only about "the pure biophysical system" but also about the human use of it (Vatn and Vedeld, 2012).

Vatn and Vedeld (2012) elaborate this point further and conclude that "fit is not only about *fitting* ecosystem dynamics, our [society's] priorities concerning these, and what rules *fit* these issues. It is also about motivations and human interaction. Therefore a theory about fit demands a theory about human motivation and choice". According to Vatn and Vedeld (2012), a possible way forward lies in the observation that human motivation is itself dependent upon institutions (Vatn, 2005, 2009). Hence, institutional structures or regimes can be designed, for instance, to facilitate or expect cooperative will and social engagement, rather than fostering people to act on their own interests (Vatn and Vedeld, 2012). This is a question of provision of incentives by institutional structures that will be discussed again in the context of interplay (see Section 3.2) and later on in the context of specific policy instruments in Section 3.4 as well.

To enhance the fit between biophysical systems and governance, coordination of institutions is a minimum requirement. Mechanisms to facilitate linkages among institutions with often autonomous but interdependent actors or actor groups are crucial to avoid fragmented and sectoral approaches (Young et al., 2008). For this purpose intermediaries, e.g. boundary organizations and bridging organizations, that establish the institutional interplay typically necessary to achieve successful fit are important (Brown, 1991; Young et al., 2008). In the context of adaptive co-management bridging organizations may provide arenas for trust building, social learning, sense making, identification of common interests, vertical and / or horizontal collaboration, and conflict resolution (Folke et al., 2005). According to Young et al. (2008), bridging organizations are "crucial for maintaining new collaboration among different stakeholder groups in order to foster innovation, generate new knowledge, and identify new opportunities for problem solving". Therefore, intermediaries such as bridging organizations can substantially avoid misfits and play a key role in social learning processes. Holzinger (2001) concludes similarly stating that "[...] disparities between functional space and political territory can only be removed by the reorganisation of political territories or by functional cooperation between the responsible jurisdictions".

Thus, advocates of adaptive IWRM are in favor of informal collaboration among multiple institutions within a river basin instead of creating a formalized, *unitary* river basin organization as generally favored in the past (Huitema et al., 2009; Borowski et al., 2008; Butterworth et al., 2010). This implies paying far more attention to the interactions among the multiple organizations affecting water use within a basin and less attention to the structure of an authority responsible for managing a river basin (Moss, 2012). Hence, Moss (2012) concludes, based on his investigations in the Wupper river basin in Germany, that problems of fit and interplay in reality are often interlinked and it may be opportune to distinguish them for analytical purposes only. Therefore, fit and interplay need to be conceived as complementary dimensions of collaborative water management.

The concept of institutional fit highlights the importance of finding the right size of a management unit based on the specific characteristics of the ecosystem to be managed. However, a prescribed size of the management unit, e.g. first order river basin, is not the best solution for implementation of IWRM in practice. Specific requirements for institutions to improve fit with river basins include the provision of mechanisms to measure and monitor the river ecosystem, make this information accessible and base decision on it. Thereby, the spatial extent of institutional mechanisms has to reflect the ecosystem properties, e.g. land use management at the level where its impact and temporal dynamics are perceived, as well as the interrelationships within the river basin, e.g. upstream-downstream dependence or by the integration of spatial planning / land use and water management. Furthermore, the institutional design to address problems of fit is also related to the resource use. Thus, institutional fit has to consider ecosystem properties but also interactions with the human system, defined by the user and uses of the natural system. A context-specific and problem-oriented selection of a management unit requires information on both, systems and their institutions. Matching the ecosystem properties and interactions with resource use requires a reorganization of institutions resulting in problems of interplay. Hence, solutions to the problem of fit through the establishment of new institutions (now based on ecosystem properties, e.g. river basins) imply problems of institutional interplay between existing and new institutions and both problems have to be solved interactively. While solving problems of fit improves integration within the natural system, the integration of land and water being the most important, solutions to problems of interplay are needed with respect to interactions with the human system, facilitating collaboration among stakeholders, participation and subsidiarity of decision-making. The problem of institutional interplay is discussed in the following section.

3.2 Problems of institutional interplay

As mentioned above, a principal concern of solving problems of fit is that the replacement of existing institutions by institutions appropriate to biophysical systems will create new institutional boundary problems and potentially new mismatches (Moss, 2004b). This problem is referred to as institutional interplay implying that "the effectiveness of specific institutions often depends not only on their own features but also on their interactions with other institutions" (Young, 1999). Therefore, the objective of addressing institutional interplay is to highlight "[...] the importance of looking beyond the design of individual institutions to the ways in which institutions interact and how this interaction can shape policy delivery" (Young, 1999). Generally, a distinction can be made between the horizontal and vertical dimensions of institutional interplay.

The problems of horizontal interplay arises because institutions operate within functionally split jurisdictions although these functions are physically linked on the ground. Thus, horizontal interplay describes linkages among institutions operating at the same level of social organization. These linkages are ubiquitous in environmental governance and with an increasing number of institutions in a given social space, interactions between and among individual institutions increase exponentially (King, 1997; Young, 2002b). Often horizontal linkages, just as vertical linkages, represent side effects or unintended byproducts of institutions designed to achieve other ends. This sort of linkages is often referred to as institutional overlaps (Young, 2002b). However, sometimes these institutional overlaps are intentional in order to solve problems or enhance cooperative outcomes and thereby make distinct institutional arrangements fit together into structures that promote a common goal. In other cases, horizontal interplay may also seek competition among institutional arrangements to advance their individual institutional agendas (Young, 2002b). Hence, institutional interplay can promote both cooperative and competitive ends in conscious efforts as politics of institutional linkages.

For this reason it is important to stress that institutional interplay, both horizontal and vertical, ordinarily generates incentives to manage interactions in order to achieve joint gains or to avoid joint losses. Attaining these positive outcomes is not an easy task since interplay often implies mixed-motive situations where actor's behavior may complicate the pursuit of a common goal. Furthermore, the achieved institutional interactions may lead to opportunities for strategic behavior for those who have little or no interest in promoting the common goal (Young, 2002b). Considering the potential positive synergies of institutional interplay in maximizing social welfare, the possible downsides have to be kept in mind.

According to King (1997), institutions can "respond to institutional interplay or overlap through adaptation of one institution to another (unilateral adjustments) or mutual adjustment in which two or more institutions are designed to work together in order to optimize joint effectiveness, muster political support, or achieve some other purpose that neither can accomplish on its own". While unilateral adjustment often occurs because of vertical institutional interplay when local regimes are superimposed by national regimes or forced to adjust to them (e.g. through the creation of RBOs having the same or similar functions as local governments), mutual adjustment more often occurs horizontally, rather than vertically, since it requires the existence of a decision-making arena that can deal with both (all) institutions involved (King, 1997; Young and Underdal, 1997).

In contrast to horizontal interplay, vertical interplay in its most common form represents linkages between institutions dealing with related issues but located at adjoining levels of social organization. Interactions between federal and state or provincial institutions dealing with the management of land and natural resources, and between state or provincial and local institutional arrangements addressing matters of environmental quality or public health are prominent cases for vertical institutional interplay in the context of water resources governance (Young, 2002b).

The directions and reasons of institutional interplay can differ substantially. While the direction of institutional interplay is either vertical (e.g. from the national to the local level and vice versa) or horizontal (e.g. among different institutions of different sectors within a river basin), the reasons for institutional interplay may also differ. According to Young (1999) and earlier work of King (1997), institutional interplay can originate from a *functional linkage* as an unavoidable interdependency between different institutions, e.g. those for agriculture and environmental protection, or from a *political*, i.e. tactical, *linkage* where institutions seek integration for a common political purpose (e.g. public health and water supply institutions to improve life expectancy). A *membership linkage* is introduced by King (1997) as a third type of institutional interplay referring to components of institutions, which overlap or have an integrative function. River basin organizations are a typical example of interplay between different participating institutions linked by membership. Table 3.6 illustrates the different types of origin in institutional interplay.

Type of institutional linkage	**Description**
Functional Linkage	Functional linkages typically reflect (inter-)dependence relationships existing in the biophysical contexts or social settings
Political Linkage	Politically constructed linkages exist whenever actors decide to consider two or more institutions as part of a larger complex or package
Membership Linkage	Refers to components of institutions, which overlap or have an integrative function (e.g. River Basin Organization)

Table 3.6: Different types of institutional linkages (cf., King, 1997)

Vertical interplay primarily, although not exclusively, arises from functional interdependence, in contrast to the deliberate or intentional links associated with the politics of design and management which often become prominent in connection with efforts to solve problems arising from the effects of functional interdependence related to horizontal interplay.

Young (2002b) argues that nationally organized regimes facilitate and sometimes promote rather large-scale, consumptive, market-driven, and often unsustainable uses of targeted resources (e.g., forests, fish) in form of commodification. This is caused by the provision of "arenas in which the interests of powerful, nonresident players generally dominate the interests of small-scale, local users" (Young, 2002b). By contrast, local regimes are in favor of small-scale uses of resources that evolve over time from the experiences of resident resource users. Furthermore, local regimes are less market-driven and give higher priority to sustainability of local ecosystems (cf., Ostrom, 1990). As local and other institutional levels coexist, environmental resources may be affected substantially by cross-scale interactions (vertical interplay) between these systems.

A common problem of vertical interplay if induced from top-down, according to Young (2002b), is the proliferation of formal rights and rules that are poorly suited to local circumstances. Additionally, local exceptions and informal interpretations can contribute to practical ineffectiveness. This is also true regarding the rights and interests of various groups of stakeholders. Considering the subsidiarity principle, design and operation of institutional arrangements need to take local knowledge into account and protect the rights and interests of local stakeholders, even when they introduce mechanisms at higher levels of social organization, required to fit the dynamics of ecosystems that are regional or even international in scope (Young, 2002b). The proper level of social organization to assign management authority, thus, is very context-specific. A promising approach here can provide forms of co-management where working partnerships between local users of natural resources and (sub)national agencies with the formal authority to make decisions about human activities impacting ecosystems as well as the resources to administer management systems are established (Berkes et al., 2000). However, layered institutional structures that imply vertical interplay are needed because of the nested characteristic of ecosystems together with the structure of nation states. Cause-effect relationships of local actions may reach the regional level but can easily reach the global level, thus, spreading beyond administrative boarders (Vatn and Vedeld, 2012).

In the context of water governance, horizontal and vertical directions of interplay, such as functional, political and membership linkages, are common for different reasons. Table 3.7 illustrates these different kinds of institutional interplay and linkage with regard to the IWRM process. New formal institutions situated at the river basin level have been introduced in most countries of the world to reduce problems of fit between administrative and biophysical boundaries (Pahl-Wostl, 2009). Thereby, problems of fit have often been solved at the expense of problems of interplay (cf., Moss, 2006). As a result newly established institutions at basin scale and those organized at traditional administrative boundaries, e.g. for land use planning or agriculture, now experience problems of vertical and horizontal interplay between them that prove to be a barrier for implementing integrated management approaches and may lead to overly complex structures (Borowski et al., 2008; Pahl-Wostl, 2009).

		Functional Linkage	**Political Linkage**	**Membership Linkage**
Vertical interaction		Transposition of national water laws into municipal legislation	Cooperation between national and municipal authorities in implementing multi-level water laws	Municipal authorities fulfilling dual functions (state and self-government tasks)
Horizontal interaction	Functional	Land use, water pollution and water protection	Cooperation between water users and farmers in achieving water quality goals	Management committee serving agricultural and water management interests
	Spatial	Dependency of downstream on upstream users	Coordinating agricultural support schemes between neighboring states	Trans-boundary coordination of water management in river basins

Table 3.7: Examples of different kinds of institutional interaction and linkages in the context of a national IWRM process; based on Moss (2004b)

With regard to vertical interplay, the IWRM implementation process is, in general, induced from top-down, based on legal and institutional reforms at the national level as well as coordination and alignment of general development goals of water resources management, with implications for all policy levels. The decentralization process pursued in IWRM implementation is based on the subsidiarity principles and is intended to assign important decision-making competencies to the sub-national level. Furthermore, it is necessary to integrate all (relevant) stakeholders in the planning and decision making processes. Thus, the IWRM process implies significant vertical interplay between different policy levels, i.e. levels of social organization. In the context of IWRM, institutional interplay additionally refers to cross-sectoral cooperation and coordination of different water use sectors and others, including land uses that affect the hydrological cycle or are dependent on it (see Figure 3.3) in terms of horizontal interplay. Vertical and horizontal interplay together represent the integration within the human system.

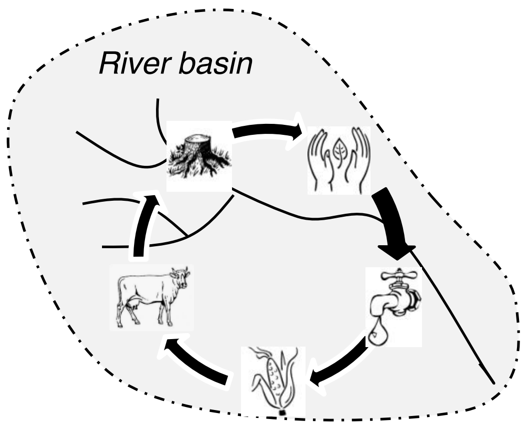

Figure 3.3: Illustration of the problem of horizontal institutional interplay within a river basin (Source: Author's work)

With regard to vertical interplay, the decentralization process of IWRM is of special importance. Decentralization offers opportunities to manage water in a context-specific integrated way and options for practical participation of local communities and other local players. Moreover, it offers more scope for timely and effective enforcement of rules (Adeyemo, 2003). But there are also significant constraints to decentralization because of a lack of funding and capacity, e.g. at the local level of municipalities. Furthermore, there is a need to develop local expertise and the introduction of integrated water management planning at district and municipality level. In case of provision of sufficient sovereignty to autonomous municipalities through an effective national enabling environment, they may be able to raise their own funds and attract domestic and even international investment for water resources management (Adeyemo, 2003). Since the present link between water management at different levels is often disjointed, conflicting or top-down it is very important to link local water management with water resource planning at river basin or national level in a bottom-up manner of vertical interplay. There is much evidence that local communities and water users can govern common resources in equitable and efficient ways (Bromley, 1992; Ostrom, 1990) while "many of the current problems of water governance derive from hierarchical and centralized control by the state and its inability to provide sufficient water-related services or to enforce regulations" (UNESCO-WWAP, 2003). Accordingly, (Adeyemo, 2003) argues that an effective governance of water resources requires the combined commitment of government and various groups in civil society particularly at local / community levels as well as the private sector. This distributed governance of water by directing more responsibility to the local and regional level is an institutional response to the IWRM principle of subsidiarity. To achieve this "confusion over the demarcation of responsibilities between and among actors, inadequate co-ordination mechanisms, jurisdictional gaps or overlaps, and the failure to match needs, responsibilities, authorities and capacities for action" has to be overcome (Adeyemo, 2003). In order to consider all interests involved, IWRM should enable full stakeholder participation at the appropriate level. Therefore, river basin plans, as the principle instrument of vertical interplay in IWRM, according to Jaspers (2003), need to be composed of lower level sub-basin, catchment or watershed plans if the scale of the river basin or specific use contexts makes them necessary. Lenton (2011) too, highlights the need to understand integration in vertical terms where "actions at one level should seek to reinforce and complement actions at other levels, within the generally agreed principle that decision making on water resources should be taken at the lowest appropriate level".

According to Moss (2004b), improvements in horizontal institutional interplay "[...] may be achieved through new 'comprehensive' institutions (involving additional transaction costs and political will and mandate) or through loosely organized cooperation agreements between existing (sectoral) institutions in form of minimal institutional change". Experience of IWRM implementation from countries with principally regulatory policies with few negotiative and participatory governance elements, e.g. Germany and Spain, suggests that the necessary institutional interplay in river basin management has posed severe problems of institutional adaptation (Moss, 2004b). Moreover, when it comes to river basin management plans legitimacy and ownership are frequent problems of institutional interplay impeding operationalization. Mitchell (2005) concludes in this context: "The result is that the IWRM recommendations often have low priority because they are perceived to be someone else's problem or responsibility. Alternatively, if implemented, they are scheduled to fit into the activities and priorities of each agency, rather than with regard to how they should be sequenced as part of an overall, integrated initiative".

It is assumed that these challenges of coordination are even greater in developing and transition countries, because these countries typically lack essential financial, institutional and human resources (Horlemann and Dombrowsky, 2011). Nielsen et al. (2013) argue that different institutional set-ups may promote or hinder synergies among institutions regulating water and related policy fields, particularly land use, as required for an IWRM. Mitchell (1990) stresses further that solving the problem of spatial fit in river basin management depends on the success in parallel improvements of institutional interplay: "[T]here is never a perfect 'fit' among legitimization instruments, functions and structures. As a result, use is made of various processes and mechanisms to overcome the problems, which occur because of imperfect matches. It is often these processes and mechanisms, informal and formal, which facilitate co-ordination and integration". Furthermore, Moss (2004b) argues that "[...] in the absence of an [appropriate] organisational blueprint for river basin management, the process of institutionalising river basin management, rather than the end-result, acquires particular significance". Moreover, he adds "[...] the task of winning broad support for a more integrated, holistic approach to water management demands extensive interaction between a wide range of parties. It requires complex negotiation and bargaining processes with other parties relevant to water resource management and the creation of new partnerships to solve basin-specific problems (from Newson, 1997)".

Moss (2006) points out that the literature on the organizational aspects of IWRM on the river basin level identifies differences in the spatial remit of institutions as a significant hindrance to cross-sectoral cooperation over water-related issues. Furthermore in this context of horizontal interplay Newson (1997), for instance, sees *policy gaps* (cf., OECD, 2011b) between land use planning and water management planning resulting from different spatial scopes of the two planning regimes, as one of the principal problems of river basin management. Hence, he concludes that IWRM cannot be achieved by institutions of water management alone because water resources are affected in quantity and quality by a wide range of human activities, e.g. agriculture, urban development, power generation etc. Hence, good institutional interplay is essential to achieve effective IWRM implementation.

Thus, there is a need to align objectives of sectoral institutions. This *objective gap* (cf., OECD, 2011b) underlines the governance challenges in fostering strategic and territorial planning of water policy. How different rationales create obstacles for adopting convergent targets can be observed at sub-national level where urban flood controls and ecological preservation or restoration of urban waters often conflict. For instance, exclusive emphasis on structural methods of flood control has led to destruction of habitat as well as deterioration of water quality in the past (OECD, 2011b). A promising way in reducing the objective gap has been found when rationales of flood control, ecological preservation and spatial planning converge (often referred to as 'issue linkage'). In doing so it is also possible to minimize the impact on other policy areas (OECD, 2011b). Cleveland et al. (1996), based on findings from a review of ecological and the social science literature, stress in this context that cooperation, e.g. in form of issue linkage, seems to be more likely where the number of actors is smaller rather than larger, where interactions are repeated, and where actors are able to detect cheating and punish offenders.

Similarly, Horlemann and Dombrowsky (2011) note that cooperation and coordination of numerous water-related organizations, if not effective, leads to inconsistent water governance in terms of problems of horizontal interplay. These problems of horizontal interplay between different water using sectors are extremely pertinent in water management because of the multi-functional character of water (Horlemann and Dombrowsky, 2011). Each sector is framed by its own institutional arrangements (cf., Moss, 2004b), thus, demands of all water users and their water-relevant institutional environment have to be balanced to resolve coordination problems and to achieve cooperation or at best collaboration. This requires coordination of possible conflicting sector institutions, e.g. economic and environmental institutions, and requires cooperation among their respective administrative bodies (Horlemann and Dombrowsky, 2011).

The concept of benefit-sharing addresses coordination and collaboration problems of horizontal interplay in moving from the sharing of water quantities to the sharing of the benefits the users receive from its use (Dombrowsky, 2010a). One reason to focus on benefits from water rather than of water quantities, i.e. allocation of water use rights, is the promise to leave the zero-sum game of water-sharing and replace it by a positive sum game of benefit-sharing (e.g. Biswas, 1999). This way potentially difficult negotiations may be avoided. Sadoff and Grey (2002) argue, for instance, that "[f]ocusing on the benefits derived from the use of water in a river basin, rather than the physical water itself, is another way to broaden the perspective of basin planners".

Type of problem	Examples	Incentives for cooperation
Provision of commonly used water infrastructure	Flood protection along shared rivers, shipping or construction of water infrastructure that benefits all parties. Cooperation includes a "production problem" in finding the most efficient project realization and a "distribution problem" concerning the distribution of cost and benefits from the project.	All parties' interests are equal in direction and cooperation is of benefit to all. Although the distribution of cost and benefits may be politically challenging, the chances for agreement are high because of the symmetric character of incentives and homogenous interests.
Cooperation in management of shared water bodies (i.e. existing water infrastructure)	Contrary to the provisioning problem of shared water infrastructure, the use or appropriation of existing water resources often has the structure of a cooperation problem (Dombrowsky, 2005) - often referred to as *prisoner's dilemma*. Individually rational decisions on resource use result in collectively irrational consequences, e.g. exploitation of a common groundwater aquifer by different parties at the same time beyond its regeneration capacity.	There is a strategic incentive problem because non-cooperation may be advantageous for some resources users if others cooperate (free rider problem). Compared to the coordination problem of commonly used infrastructures the probability for cooperation is reduced if collaboration cannot be supported through positive or negative incentives.
Unidirectional externalities (upstream-downstream)	Classical problem constellations within river basins where upstream activities lead to positive or negative externalities downstream. Possibilities for collaboration are limited because individual actors are not interested in cooperative solutions because incentives are missing. E.g. pollution control or flood protection measures where only downstream populations benefit.	Incentives for cooperation are lowest because of asymmetrical interests. Cooperation can only be expected if compensations for opportunity costs are paid.

Table 3.8: Typology of coordination and collaboration problems in IWRM; based on Klaphake (2005); Dombrowsky (2005)

The need for benefit-sharing arises because not all actors influencing the availability and quality of water resources agree on a common water, i.e. land use for optimal water management (cf., Rogers, 1997; Marty, 2001; Dombrowsky, 2005). This coordination problem can be further differentiated in three types of coordination, i.e. collaboration problems (see Table 3.8):

1. Provision of common infrastructure
2. Cooperation in (transboundary) management of commonly used water bodies
3. Unidirectional externalities (upstream-downstream)

According to Dombrowsky (2005), a distinction can be made between coordination and cooperation problems in water resources management depending on the respective problem structure. A typical cooperation problem between different parties (in water resources management the provision of commonly used water infrastructure has usually the structure of a coordination problem) contains a production problem and a distribution problem. On the one hand, the production problem consists of determining which measure results in the highest welfare improvement. On the other hand, the distribution problem consists of finding an agreement on the distribution of (Pareto-optimum) net benefits among all parties. The production problem is purely a coordination problem whereas the distribution problem represents a zero sum game where gains of one party imply losses of another party. If an agreement on the production function is achieved, all parties have an interest in realizing the agreement. This solution to a coordination problem is therefore called a self-enforcing agreement (Barrett, 2003).

Contrary to the above mentioned provisioning problem of shared water infrastructure, the use or appropriation of existing water resources often has the structure of a cooperation problem (Dombrowsky, 2005), often referred to as *prisoner's dilemma*. In this kind of problem, individually rational decisions on resource use result in collectively irrational consequences. One example is the exploitation of a common groundwater aquifer by different parties at the same time beyond its regeneration capacity.

Both, the provision of common infrastructure and the cooperation in the management of commonly used water bodies, are also referred to as reciprocal externality problems (cf., Dombrowsky, 2007a). In most cases, however, externalities are unidirectional, due to the flow direction of water, leading to typical upstream-downstream problem constellation. A unidirectional externality rules out reciprocal effects in the same use. According to Dombrowsky (2007a), unidirectional externalities, whether positive or negative, rule out reciprocal effects in the same use. Figure 3.4 illustrates different types of unidirectional externality problems in water management and provides typical examples for each type.

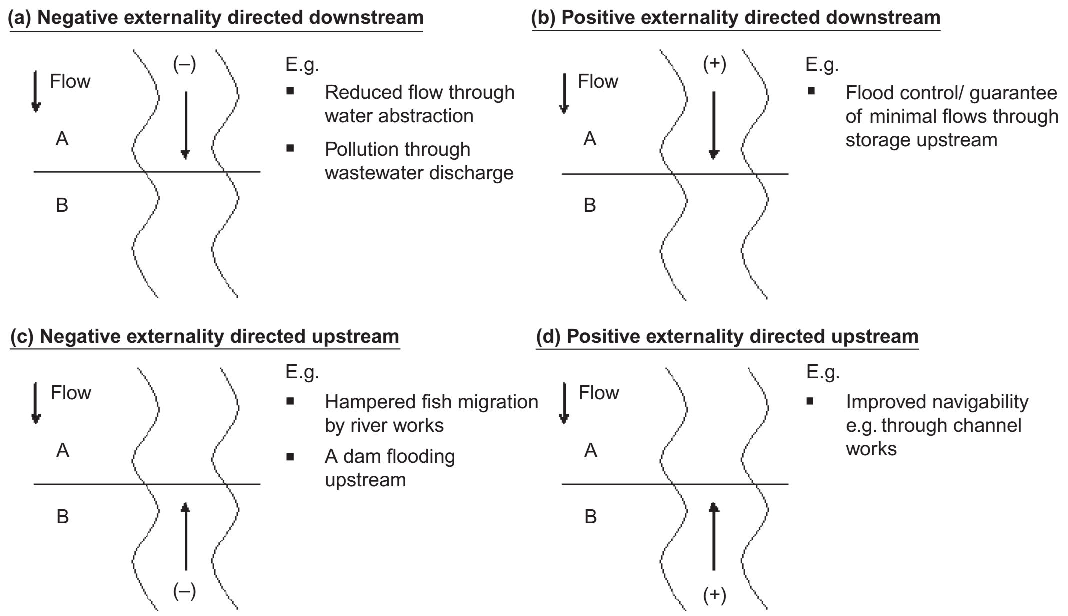

Figure 3.4: A typology of unidirectional externality problems in transboundary water management (Dombrowsky, 2010b)

The unidirectional character of problem constellations is further typical with regard to the impact of land use on water resources. In this regard, upstream-downstream relationships are a prominent example of functional linkage requiring an integration of land and water institutions to be addressed properly. IWRM usually attempts a political linkage of the sectors, however, a political linkage on its own has proven to be widely ineffective. Watson (2004) refers to coordination as an arrangement of two or more institutions that create and / or use existing decision rules in order to align their separate policies, programs or practices (cf., Mulford and Rogers, 1982). A formal, rigid and rule-based relationship is implied which enables assisting each other in working towards their separate objectives. Watson (2004) stresses that coordination is necessary to overcome problems of administrative fragmentation, overlap and duplication in river basin management, but on its own does not provide adequate institutional capacity to deal with land and water management problems that are often characterized by complexity, change, uncertainty and conflict. Hence, "co-ordination alone does not represent an adequate institutional strategy for IRBM" (Watson, 2004). He criticizes that inter-organizational coordination in IRBM is a 'myth' that needs to be reformed. In order to be successful, collaborative approaches to IRBM are necessary that ultimately depend on the design of institutional arrangements and the ability of participants to reach consensus through effective negotiation (Watson, 2004). Intractable problems, Watson (2004) notes diffuse pollution from agriculture and the loss of wetlands and biodiversity among others, usually "transcend the interests and jurisdictions [...] operating in different policy arenas, at different scales and with different understandings, values, attitudes and beliefs regarding the use and management of land and water" (Watson, 2004). Since effective solutions are beyond the reach of any single institution or actor, in such circumstances "producing agreement over causes, consequences and management responses for the problem is extremely difficult" (Watson, 2004).

Consequently, based on experiences and insights regarding the key institutional conditions and arrangements in the Fraser Basin, Canada, Watson (2004) elaborates key features or design principles that are required for a collaborative approach to IRBM to be successful. This approach is described by the acronym *CARIBOO* (see Table 3.9).

Key feature	Description
Common Vision	of the desired future conditions in the river basin and the principles that will guide actions towards realising those conditions
Adaptive Capacity	to enable policies and practices to be adjusted as new knowledge emerges and as conditions change
Resources	to enable the collaborative arrangement to function effectively and to progress from problem setting to direction setting through to implementation and monitoring
Independence	from government control, but with continued government involvement in decision-making. Though, replacement or duplication of the functions of existing government organizations involved in land and water management is not attempted
Balance	to enable diverse groups with varied economic, social and environmental interests to be fairly represented
Outputs	to ensure the arrangement is action-oriented and not just a forum for debate
Outcomes	to demonstrate that collaborative efforts have a positive impact on the sustainability of the river basin systems

Table 3.9: The CARIBOO model for collaborative IWRM at the river basin level; from Watson (2004)

Watson (2004) argues that a collaborative governance model offers a number of potential benefits for IRBM, which cannot be provided by conventional top-down or even highly coordinated institutional arrangements. Especially in the context of "edge and boundary problems" of overlapping powers, duties, jurisdictions and interests of several institutions, resulting from solving the problem of fit, a collaborative approach can provide a mechanism to deal with them in a joined policy process (Mitchell, 1990; Watson, 2004). Nevertheless, for Watson (2004) collaboration in IRBM is unlikely to be a smooth or conflict-free process. Thus, sustained negotiation and facilitation will be needed in order to engage the interested parties.

Hence, a special concern of horizontal institutional interplay is related to functional, i.e. physical, linkages between different water and land users. These functional linkages are difficult to resolve. Dixon (1997) stresses that in many countries, in the context of water resources management, there is a marked difference in the location of political as well as financial power and the relative flows of benefits between groups inhabiting the upper watershed and those who live further downstream. Based on a case study from Asia, Dani (1986) explored these interactions and identified typical upland-lowland patterns of decision-making, resource inputs, and flows of benefits. Figure 3.5 illustrates these patterns of political and social interactions between the lowland and upland inhabitants in a watershed that often benefit the lowland communities more. Upper watersheds, according to Magrath and Doolette (1990), apart from being physically remote, are often politically remote from decision-making as well. This uneven distribution of financial and political power, as shown in Figure 3.5, results in an uneven division of the benefits from improved watershed management. This is because the resources required to implement the policies are disproportionately paid by the upland group while a disproportionate

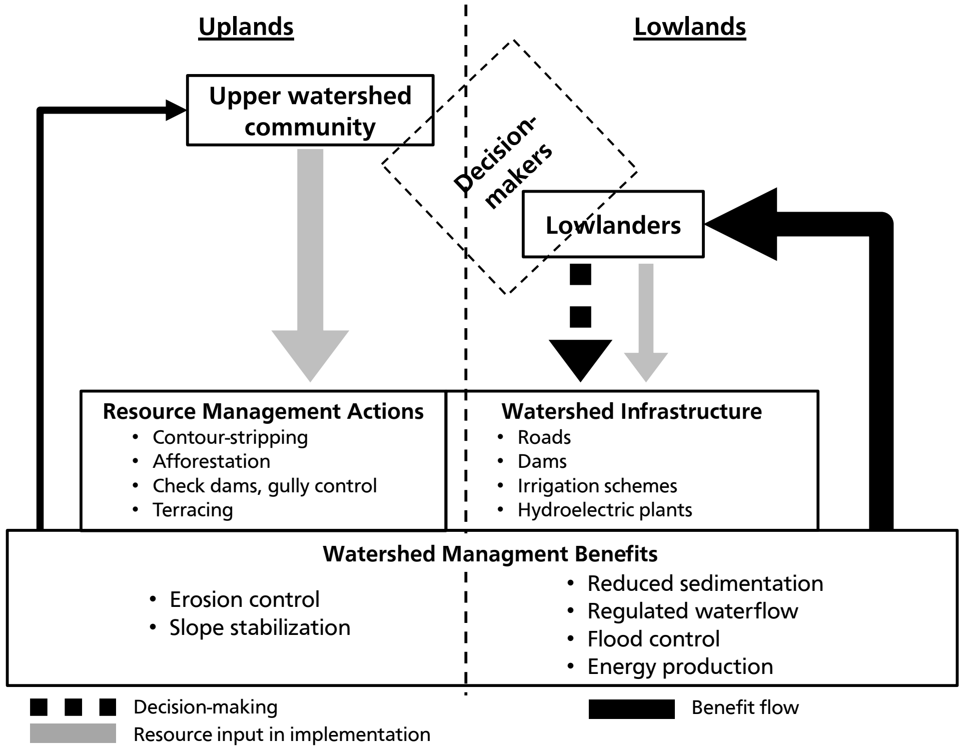

Figure 3.5: Highland-lowland relationships in watershed management; based on Dani (1986) as published in Dixon (1997)

share of the benefits goes to the lowland group. According to Dixon (1997), "this separation of benefits and costs [is] not surprising given that the main opportunities to conserve soil are in the upper watershed while a larger share of population and infrastructure are located in the lower part of the watershed". Dixon regards the underdevelopment of the upland group as one result of this process which is frequently accompanied by political and social tensions, and an 'us-them' attitude.

According to Dixon (1997), the interactions among up- and downstream stakeholders as well as the distribution of cost and benefits identified by Dani (1986) may explain the rather unsuccessful record of many watershed-management projects in developing countries. Those who are asked to implement conservation practices, e.g. different cultivation or land-management practices, often do not see themselves as the beneficiaries of these changes (Dixon, 1997). In contrast, downstream decision-makers, which are negatively affected by existing upstream cultivation or land-management patterns, "see upland residents as a difficult, somewhat foreign group that needs to be *managed*" (Dixon, 1997). Effective resource management and cooperation / communication among stakeholders, thus, are obviously negatively affected by these tensions. Mostert (1999) agrees that such asymmetric power-relations caused by hydrological factors are a principal characteristic of IRBM. Regardless of political power distribution, upstream users depend less on the downstream users than vice versa. On its own, this unidirectional dependence results in fewer incentives to co-operate for upstream users than for downstream users (cf., LeMarquand, 1977; Marty, 1997). For some uses, e.g. shipping, the dependence may also be of reverse direction and, more importantly, relations resulting from hydrological factors are only one aspect of the relationship between the upstream and downstream users (Mostert, 1999).

This complex stakeholder interaction illustrates that the reduction of misfit by the introduction of hydrological boundaries defining the area of management can lead to greater challenges with regard to collaboration and participation of stakeholders. These problems have also been documented in the context of implementation of the WFD in the EU (Moss, 2004b; Mostert et al., 2007; Timmerman et al., 2008). Mostert et al. (2007), based on the examination of WFD implementation experiences in 10 European river basins, conclude that stakeholder participation is often only consultative without decision-making features. He describes the situation in the 10 European river basins as follows: "Quite often, the existing governance style was not participatory, and it took a lot of convincing to move toward multiparty collaboration. In many cases, the authorities lacked experience with multiparty approaches, relied heavily on technical expertise, feared to lose control, or feared that too broad participation could threaten the confidentiality of the proceedings. As a result, participation often remained limited." The OECD (2011b) refers to *accountability gap* when insufficient users' commitment as well as lack of concern, awareness and participation are present (see Table 3.1 in the introduction of this chapter). A shortening of the decision-making process, in particular when local governments do not have the capacity to monitor and civil society is not totally engaged, introduces risks of transparency, integrity, capture and corruption (OECD, 2011b).

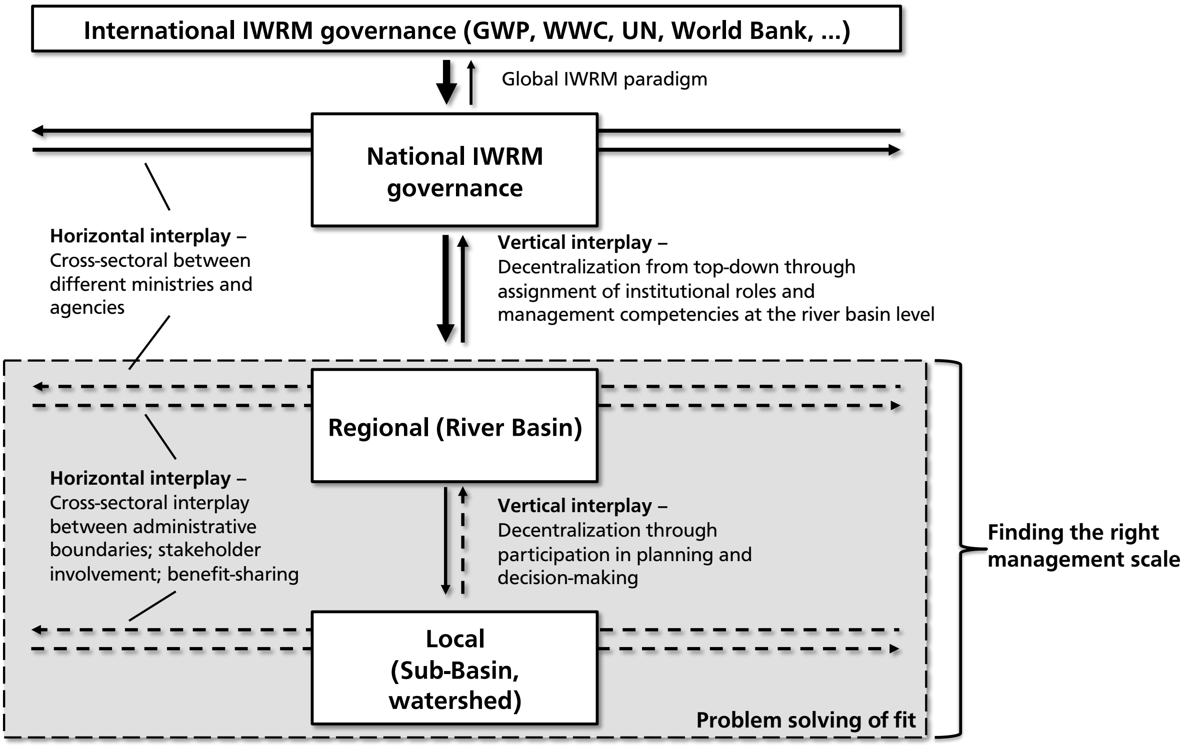

Figure 3.6: Interaction of issues of institutional interplay and fit in a national IWRM implementation context (Author's work)

The main findings of this policy analysis concerning spatial fit and institutional interplay in the context of IWRM are the interdependence between creating an appropriate managing unit that reflects the characteristics of the natural resource to be managed (problem of fit) and a successful cooperation as well as coordination between different institutions (problem of interplay). This interdependence is very site-specific, thus, creating new spatial areas of governance and new institutions from top-down remains a challenge (Moss, 2004a). This circumstance may explain why prescribed solutions from top-down for problems of spatial fit and institutional interplay do not work well, especially in an environment where existing institutions are weak and lack resources as for instance in many developing countries. While using traditional 'command and control' policies, usually bound to respective administrative limits, there is no incentive, neither for coordinated action to improve spatial fit, nor for cooperation in terms of improved institutional interplay (Serageldin, 1995). However, an OECD survey on water governance revealed that "more than two-thirds of OECD countries surveyed explain the remaining policy gap by the lack of institutional incentives for encouraging inter-institutional co-operation at horizontal level" (OECD, 2011b). Mitchell (2005) describes this separation of responsibilities among resource-management institutions and their inability or unwillingness to consider their mandate relative to those of other organizations as "silo effect" (cf., Serageldin, 1995).

Figure 3.6 illustrates the interdependence of different forms of institutional interplay and the problem of fit in the context of a typical national IWRM implementation process. The problems of institutional fit and interplay have to be solved interdependently in order to find the right scale for management corresponding to prevailing natural and human system's characteristics. These principal challenges of operationalization of IWRM are illustrated in the figure with a grey shaded box.

3.3 Principal problems of operationalization in the context of IWRM implementation in developing countries

The implementation of IWRM is characterized by specific information, capacity and funding gaps which generally have their origin in a typical resource scarce environment of these countries (OECD, 2011b; Akhmouch, 2012). Besides the problems of fit and interplay as discussed in Sections 3.1 and 3.2, these implementation gaps represent significant additional obstacles which have to be considered when choosing instruments for the operationalization of IWRM.

One of the principal problems of IWRM operationalization concerns available information on the human and natural system to be managed. On the one hand information is not shared because of sectoral fragmentation of water-related tasks across levels of government and local actors involved in water policy, and on the other hand information often does not exist at all. This lack of information is a primary concern regarding guidance for decision makers. Although in practice sub-national, i.e. local governments will tend to have more information about local needs and preferences, and also about the implementation and costs of local policies, this information is often not considered in higher-level decision-making (OECD, 2011b; Akhmouch, 2012).

While one aspect of the information gap is that only limited information is available on the human and natural system. Another aspect relates to the way how the rationale for an integrated management of water resources is communicated among different stakeholders. In case of a widely-held perception of the depletion or degradation of water and natural resources, the implementation of IWRM is more likely to occur. Thus, the information gap is also about problem awareness and willingness to act on the part of government authorities and water users to invest financial resources and labor in water management, rather than in another area of public concern. Lee (2000) stresses that, as far as cities in developing countries (especially in Latin America) as the largest single water consumers and polluters are concerned, there has been little history of cooperation among public authorities, city residents, and the private sector concerning river basin management and stewardship for this task has been neglected. Although land use regulations and environmental protection laws designed to prevent water resource deterioration often exist on paper, these are not applied due to a lack of will and / or the necessary resources for monitoring and enforcement (Lee, 2000). This holds also true for the implementation of water management plans which many of the least-developed countries are unable to achieve because of lacking financial, managerial, and political capacity (Lee, 2000).

Besides an informational gap there is evidently also often a capacity gap when local authority's organizational, technical, procedural, networking and infrastructure capacity is not able to assume additional water responsibilities from implementation of national water policies (OECD, 2011b). Hence, the fundamental problem of the decentralization process in IWRM is at which sub-national level responsibilities should be assumed and decentralized tasks be executed. There is no general answer to this question. The OECD (2011b) argues in its assessment on IWRM implementation gaps that, on the one hand, there is often more capacity with regard to strategic or normative decision-making and accessibility to funding and, on the other hand, more specific knowledge at the local level, e.g. in estimation of opportunity or transaction costs for local actions. Therefore, according to the OECD (2011b) there is a need for instruments to build local capacity and for blending of knowledge of multi-level governance.

Another important operationalization obstacle refers to insufficient or unstable revenues to implement water policies across levels of government creating a funding gap. The decentralization process often results in delegation of responsibilities to lower administrative levels but without assignment of additional funds to fulfill these tasks (OECD, 2011b; Akhmouch, 2012). However, decentralization is necessary in order to bring decision making closer to those affected by governance, thereby promoting higher participation and accountability; and, it can help decision makers take advantage of more precise time- and place-specific knowledge about natural resources (Lemos and Agrawal, 2006). The lack of capacity and information is somehow an expression of a lack of adequate funding. Moreover, the financing of IRBM tasks is broadly neglected and there are no revenues available to cover the costs. As a result, private partners, investment banks and innovative arrangements at local level have been explored as complements to public action in water financing (OECD, 2011b; Akhmouch, 2012). The funding gap often causes a dependence of sub-national authorities on higher levels of government for funding water policies, while central government depends on the sub-national authorities to deliver them and meet both national and sub-national policy priorities (Akhmouch, 2012). Given these funding constraints, the OECD (2011b) stresses the need for shared financing mechanisms between the principal actors involved, e.g. the national and sub-national authorities as well as the private sector.

The funding gap is closely related to the administrative gap. In river basins shared by different jurisdictions (e.g. municipalities) insufficient coordination and collaboration of public authorities often results in a lack of an integrated approach and territorial customized water policy that compromises the efficiency of budget execution (OECD, 2011b).

In developing countries, the main constraints to IWRM or policy implementation in general are strongly related to a weak performance of government at all levels. However, this is due to limited resources in terms of capital, personnel capacity and information on environmental resources in order to enforce strong interventionist policies. Moreover, corruption and political arbitrariness have led to very low confidence in the government as a constructive actor to solve environmental problems. This explains the tendency, in many developing countries, of informal rules to override formal rules, making the enforcement of formal rules even more difficult and thereby affecting performance (Bandaragoda, 2000). These informal *rules-in-use* become dysfunctional when they contradict formal rules or replace formal rules that become ineffective due to a lack of proper enforcement. Another important aspect that affects the success of regulatory government interventions is the dependence of smallholders on (marginal) environmental resources for subsistence and their limited capacity, e.g. because of lack of knowledge or opportunities, to tap alternative sources of income.

Horlemann and Dombrowsky (2011) studied the national IWRM implementation of Mongolia. One of their major findings was that, "as elsewhere, the decentralisation process contributed to problems of fit in the sense that strengthened jurisdictions at lower levels now face the challenge to coordinate at river basin level. In addition, it amplified problems of vertical institutional interplay, in particular due to its incompleteness and an unclear allocation of decision-making and fiscal competences" (Horlemann and Dombrowsky, 2011). Nevertheless, Horlemann and Dombrowsky (2011) stress that in the Mongolian case steps had also been taken which potentially address these problems of fit and interplay. The most important step is identified with the establishment of River Basin Councils (RBC) which are supposed to balance local water user interests by coordinating different sectoral interests and different administrative levels at the basin scale, thereby, addressing problems of fit, horizontal and vertical interplay simultaneously. According to Horlemann and Dombrowsky (2011), the lack of implementation capacity and financial resources remains a huge constraint for the Mongolian River Basin Council (RBC)s and can only be solved through associated River Basin Authorities (RBA) or the establishment of water administrations at provincial and district levels. Additionally, the political decentralization has to follow a fiscal decentralization in order to generate the necessary resources for IWRM implementation (Horlemann and Dombrowsky, 2011). Hence, specific organizations are established to solve prevailing problems of interplay (e.g. the Mongolian National Water authority to tackle vertical interplay at different administrative levels and the National Water Committee to foster cooperation of the water-related ministries addressing horizontal interplay), but "vested interests by line ministries, long-established institutions and strong sectoral segregation resist efforts to better institutional interplay". The success of such newly established institutions is further diminished by endowment of few resources for enforcement for their tasks. Thus, Horlemann and Dombrowsky (2011) consider the allocation of decision-making and financing competences, not only for Mongolia but in general as a challenge in water resource management.

Horlemann and Dombrowsky (2011) regard the analytical framework of fit and interplay introduced by Young "as valuable tools to structure the analysis of institutional arrangements conducive and obstructive to an IWRM". But similar to (Moss, 2004b) they also highlight the difficulty of regarding the institutionalization of IWRM exclusively as a problem of fit or vertical or horizontal interplay. Issues of fit and interplay, thus, are viewed as interdependent. Moreover, Horlemann and Dombrowsky (2011) argue that the success or failure of IWRM implementation cannot be reasoned sufficiently by issues of fit and interplay alone. Because of the political dimension of IWRM (cf., Allan, 2003b; Mollinga, 2008; Biswas, 2008) they recommend adding actor-centered explanatory approaches to the fit-interplay concept (Saravanan et al., 2009; Moss and Newig, 2010). Cardwell et al. (2009) sees the actor-perspective also as crucial when he argues that the scope and degree of which organizations to involve and how to involve them depends on specific actions which may significantly affect them or from which they are affected. In either case activities should be integrated or at least coordinated.

The need to solve problems of fit and interplay as well as the operationalization constraints specific to developing countries have several implications with regard to the suitability of policy instruments. It seems necessary to find appropriate instruments, complementary to existing ones, which include some kind of priority setting in order to improve IWRM implementation. Agyenim and Gupta (2012), therefore, argue that countries with limited resources should aim to define what needs to be done first and what can be postponed. This way it can be identified how existing resources can be focused and concentrated to address the priority goals first, rather than debating on how to get the institutions and policy planning process right first. Moreover, instruments that provide incentives for active stakeholder involvement, cooperation and public participation may possibly reduce the costs for specific actions to achieve this.

The following section analyzes the suitability of different policy instruments in addressing the problems of institutional fit and interplay in the presence of operational constraints. A focus is laid on the incentives provided by the instruments.

3.4 Instruments for IWRM implementation and operationalization

The problems of fit and interplay, as described in Sections 3.1 and 3.2, arise as additional governance challenges from the implementation of IWRM. In order to solve these problems, complementary policy instruments are required that consider the ecosystem characteristics of river basins and the interaction of relevant stakeholders across sectors and administrative boundaries. Both of these problems have to be addressed in a context-specific manner, this means that the physical characteristics of ecosystems and the actors involved in their use have to be addressed. The aim is to reach the appropriate scale for management, i.e. operationalization of IWRM.

Addressing the problems of fit and interplay has not been a specific subject of sectoral policy instruments in the past. Fit was predefined through administrative units, thus, no specific instruments were necessary to achieve appropriate fit. Institutional interplay has been avoided through sectoral policies covering main political sectors according to jurisdictional responsibilities. Hence, policy instruments where the legislative authority is the main actor in implementation are bound to administrative boundaries and often also to specific policy sectors. Vatn (2010) stresses this point by arguing that traditional environmental regimes have not been focused on managing interdependence, instead they have been divided according to individual resources (e.g. forests, natural or agricultural areas) or uses (e.g. forestry, agriculture, nature conservation). Accordingly, this construction of separated management units has led to great institutional challenges (Vatn, 2010). On the one hand interactions across different management units have created coordination problems, and on the other hand, there are significant externalities across the boundaries of administrative management units (Vatn, 2010).

This has changed in the context of IWRM with the insight that water resources ought to be managed according to natural boundaries of river basins and in an integrated, cross-sectoral manner. The dominant instruments of IWRM implementation today are: Integrated River Basin Management, River Basin Organizations (RBO) and River Basin Management Plans.

Integrated River Basin Management and RBOs are the prevailing attempts to address the governance challenges of institutional fit and interplay by defining the scale for management and the actors involved. The elaboration of River Basin Management Plans is the most common form of operationalization of IWRM.

Integrated River Basin Management defines the river basin as the principal unit for management. Besides first order river basins, river sub-basins are often also recommended as management units. This solution to the problem of fit is commonly applied from top-down by a national water authority, ministry or specific water laws. According to Dombrowsky (2007b) an organizational solution and a cooperative solution are possible to solve problems of fit between hydrological and administrative boundaries (Oates, 1999; LAWA, 2003; Dombrowsky, 2005). In case of an organizational solution a specific River Basin Organization is created and responsibilities are transferred from the existing political-administrative units, e.g. states or municipalities, to the river basin organization. This kind of newly established organization requires its own budget and also the power to set norms.

In contrast, a cooperative solution, e.g. a cooperative structure like a working group at the river basin level, represents an organizational arrangement of existing relevant jurisdictions to facilitate cooperation and to reach agreements. In this case, however, the budgetary and norm-setting competences remain with the respective jurisdictions. Thus, establishing the river basin as a management unit implies creating new structures, changing roles and responsibilities to meet the goals of IWRM. This is a complex task and there is evidence that the introduction of new organizational structures such as river basin organizations has been disappointing in many countries (Cap-Net, 2008). Moreover, there is often uncertainty about the role and functions of river basin organizations in the context of IWRM implementation.

The shift from traditional water resource management based on administrative boundaries to a cooperative solution is smaller than a shift to a true organizational solution. The coordinated solution can be regarded as in between the organizational and the traditional administrative solution. The coordinating arrangements in cooperative solutions have, in principal, a coordinating task, as extensive river basin planning, under most organizational solutions, often does not exist (Cap-Net, 2008). However, in most developing countries organizational solutions through the establishment of RBOs are encountered, often at the strategic level of first order river basins without operational solutions at lower levels. Moreover, these prevailing approaches to solve the problem of fit rely on mandating the management of water resources according to river basins from top-down in a regulatory manner. The creation of organizational or cooperative institutions is an attempt to create membership, i.e. political linkages among involved sectors and administrations (see Section 3.2). The expectation is to achieve management on the basis of river basins (cross-jurisdictional cooperation) and cross-sectoral collaboration, but direct incentives towards relevant actors to do so are actually missing. The experience so far has shown that mandatory and comprehensive integration from top-down has not led to the expected results. Instruments are needed to facilitate context-specific integration based on solving the problems of fit and interplay interactively. According to Moss (2006), empirical studies of experiences with river basin management indicate that a flexible and contextually sensitive approach can help to solve problems of fit and interplay more interactively.

Based on the definition of river basins as management units, the resulting problem of interplay is addressed through the creation of organizational or cooperative entities composed of predefined sector representatives as cross-sectoral structures. However, these approaches to address the problems of fit and interplay are realized in a consecutive manner based on the natural and human system's knowledge at the national, i.e. normative or strategic level of IWRM implementation (cf., Mitchell, 1990). The interdependence of institutional fit and interplay, as discussed in Sections 3.1 and 3.2, is not specifically considered in this *standard* process. As a result, the determined governance scale is not based on context-specific human and natural system characteristics. Instead it is determined on the basis of apparently obvious hydrological characteristics, e.g. the size and order of a river basin. The additional inclusion of smaller governance scales, e.g. river sub-basins, follows the same logic. Moreover, in developing countries, e.g. as documented by Horlemann and Dombrowsky (2011) for the case of Mongolia, lower level instances are often lacking because of insufficient resources.

Cap-Net (2008) studied the performance of four RBOs in developing countries (Sri Lanka, Kenya, Mexico and Malaysia) with regard to the legal framework under which the respective RBO operates, its level of autonomy and effectiveness (comparing objectives / performance targets with the actual workings), the involvement of stakeholders and financing of RBO activities. The study revealed, in three of the cases, that the existing legal framework was not limiting the performance of the RBO, only in one case the legal framework provided limited opportunities to involve stakeholders in the decision-making process. The level of autonomy of all RBOs was limited by the state government because approval for some decisions needs to be obtained from national agencies and much of the RBOs funding goes through a national agency. For all cases this political influence at some point was regarded as being detrimental to the functioning of the respective RBO. The objectives of all organizations are directed towards a holistic approach to water management but the RBOs examined are actually unable to achieve these ambitious objectives. The actual activities of the RBOs differ from these objectives due to limited human, financial and institutional capacity of the organizations. Since the organizations do not have the required resources to address all the water management tasks at their disposal they must select priorities on which to focus. Therefore, the study documents that the RBOs eventually decide to focus on the problems that are most acute or are given the highest priority. This priority setting depends strongly on problem perception at the level of the RBOs decision-making (Cap-Net, 2008). In all RBOs examined, mechanisms for stakeholder participation could be established, but this does not guarantee actual participation which in practice was very limited. Financing of RBO activities is limited to scarce financial resources and there is a strong dependence on transfers from central governments and the donor community. In cases where RBOs are able to raise funds this income is transferred to general government budgets rather than being available to the RBO for financing its activities. Moreover, the lack of financial resources has hampered the design of effective mechanisms, especially for environmental protection, in the Lerma-Chapala-Santiago basin, Mexico, (Cap-Net, 2008).

Thus, this approach to solve the problems of fit and interplay may not achieve the most appropriate scale to effectively address specific human-natural system interactions. For the human system context, this can result in involving stakeholders that do not share common problems or benefits at the determined governance scale and consequently do not have incentives for cooperation or collaboration. In the context of the natural system, this can mean that important ecosystem properties are not captured. Another problem of this approach of defining the governance scale stems from its dominant top-down orientation without a complementary bottom-up counterpart accounting for appropriate stakeholder involvement and public participation. The actors to be involved in IRBM and RBOs depend on the determined governance scale without consideration of specific contexts. Although public participation and stakeholder involvement is often encouraged in this form of IWRM implementation, the applied mechanisms often have a *public hearing* character without consideration of possible functional connections between the actors involved. Thus, it can be concluded that there is still room for improvement in finding the right governance scale.

A hierarchical structure of RBOs and RBCs is reasonable in order to address strategic and operational issues at different governance levels (cf., Mitchell, 1990). However, success in establishing lower level instances at the operational level has proven difficult and has generally been based on top-down decisions. Hence, instruments promoting a context-specific identification of the right operational scale for management from bottom-up are missing. This is illustrated schematically in the upper part of Figure 3.7. The left hand side of the figure describes the methodological conceptualization of the core problems of IWRM implementation and operationalization, the right hand side illustrates the dominant conceptual application in practice. The dashed arrows and question marks indicate critical issues of dominant IWRM implementation practice.

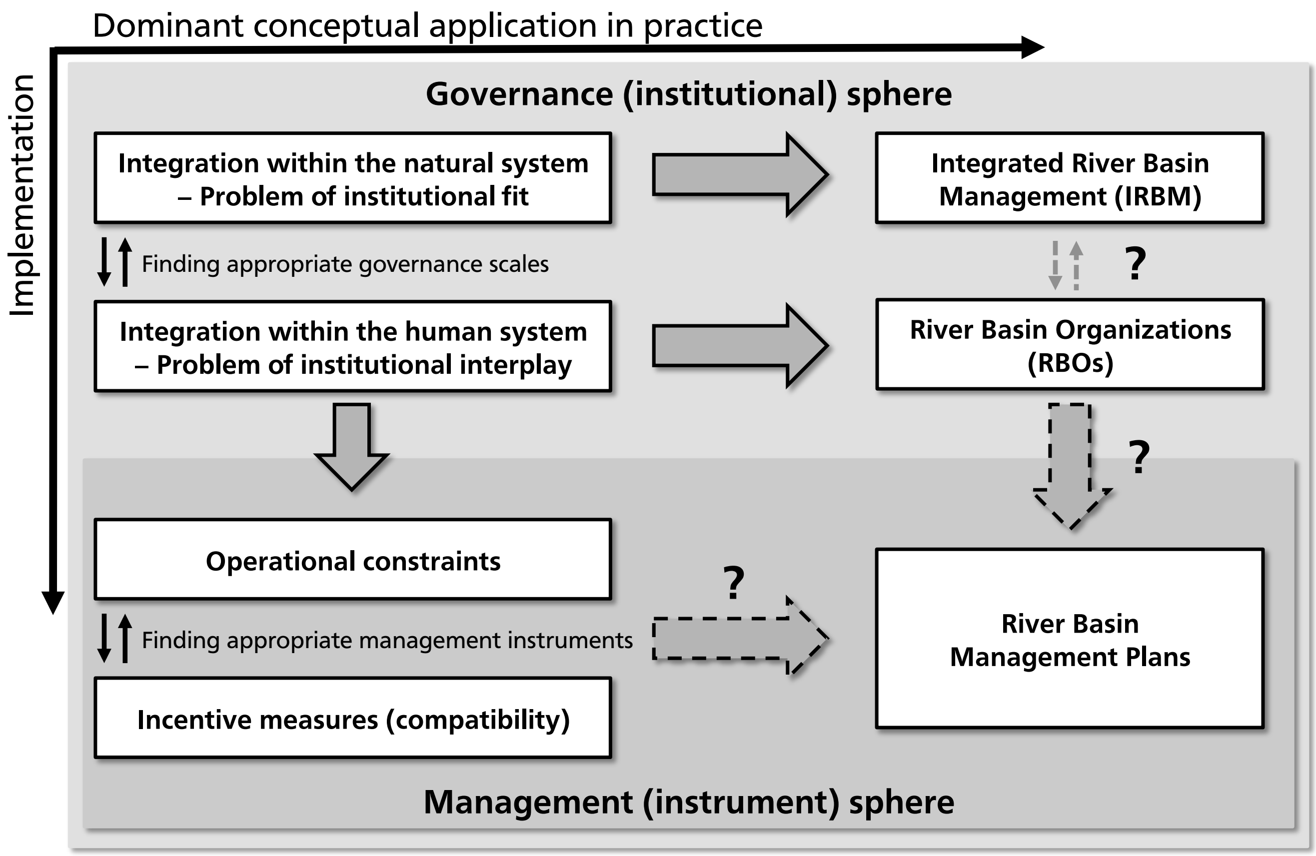

Figure 3.7: Conceptual methodological application and implementation in practice (Author's work)

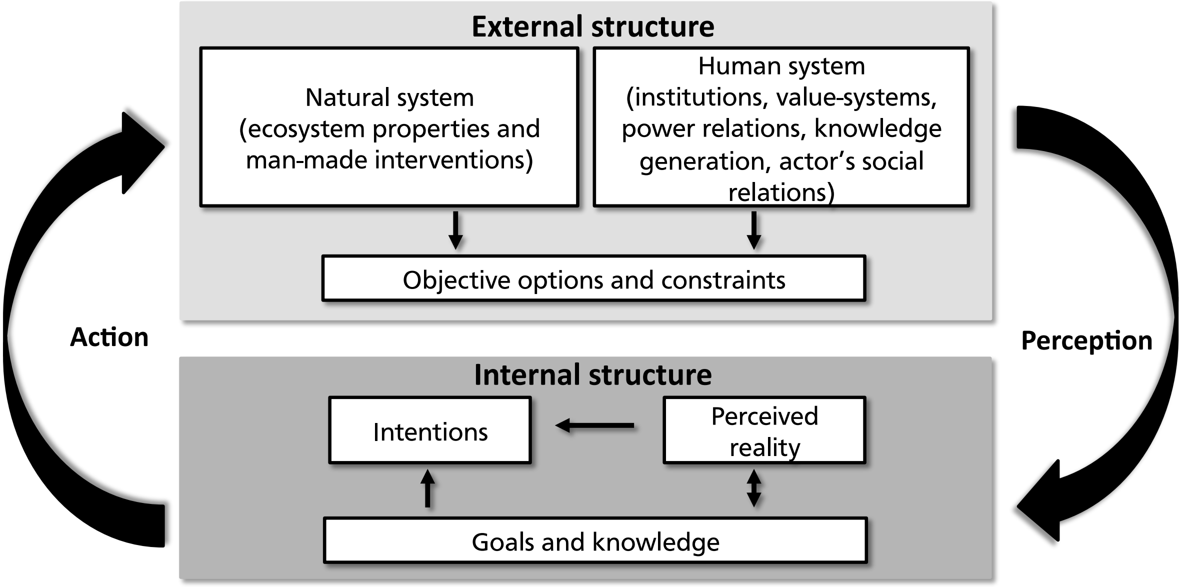

Figure 3.8: Simplified model of human and organizational action; based on Kaufmann-Hayoz and Gutscher (2001)

Besides finding the right governance scale, there is also the need to find appropriate management instruments that provide incentives to put the IWRM principles into practice despite the presence of operational constraints. This implies land use planning and related policies at the determined scale of governance, involving all relevant stakeholders in achieving cross-sectoral coordination and cooperation in order to manage water resources in a sustainable way. An actor-oriented consideration of stakeholder interaction, especially in integrating land and water management issues, is important.

Environmental management instruments are often rated according to their ecological effectiveness and economic efficiency. Ecological effectiveness includes an instruments accuracy in meeting an ecological goal and also the duration until a desired impact is achieved. Cost-effectiveness, promotion of innovations and impact on existing structures are elements of economic efficiency (Michaelis, 1996). Cost-effectiveness considers opportunity costs of other policy instruments to achieve the same goal as well as transaction and implementation costs of the policy instrument itself. Concerns about equity issues of policy instruments in terms of their impact on burdens and gains of different actors affected have also been considered as evaluation criteria. Furthermore, institutional preconditions for instrument implementation and related practicability of instruments have also been a reason for consideration of policy instruments (Grunewald and Bastian, 2012). In the context of this dissertation, the aim is to identify complementary instruments that are able to address the specific problems of IWRM implementation, i.e. operationalization of fit and interplay. The ecological effectiveness and economic efficiency of possible instruments is not directly a subject of this dissertation, although they will be addressed in the following chapters from time to time. The conclusion of IWRM implementation assessments so far is that important incentives to operationalize IWRM are not provided by the most common instruments in use, therefore, these instruments fail in achieving IWRM implementation at the operational level in a context specific manner (Bellamy et al., 2002). Thus, the following discussion of management instruments focuses on the capacity of certain instruments to address potential stakeholders directly, encouraging participation from bottom-up, raise additional funds and address context-specific problems of local importance.

A suitable way to look at different policy instruments to achieve these requirements, is from an incentive perspective of behavioral change of the actors involved (cf., Kaufmann-Hayoz and Gutscher, 2001; Börner and Vosti, 2013). Following the definition of Kaufmann-Hayoz and Gutscher (2001) instruments are meant as a goal-directed influence of an actor upon the conditions that determine a target group's action. Kaufmann-Hayoz and Gutscher (2001) present a simplified model to explain human as well as organizational action composed of an interdependent internal and external structure (see Figure 3.8). The internal structure includes goals and knowledge which result in intentions. However, goals and knowledge as well as derived intentions are influenced by the reality perceived. These factors together result in a certain action that addresses the external structure composed of the natural system (ecosystem properties and man-made interventions) and the human system (institutions, value-systems, power relations, knowledge generation, actor's social relations). These two systems define objective options and constraints which in turn are perceived by the actor's internal structure. Policy instruments address external and internal structures in different ways, thus, incentivizing actor's behavior in different ways as well.

A concrete realization of an instrument is described with the term *measure*. Different instrument types are characterized with respect to their rationale (mechanism), actors to whom the instruments are available, their target group(s) (whose behavior is intended to be influenced) and implementation and enforcement requirements. In this context a special focus is laid on how the different instruments address specific contexts with regard to institutional fit and interplay, respond to operationalization constraints and embed in existing institutional frameworks. Furthermore, an additional concern is how the instruments influence actor's behavior and how actor's decisions are influenced. In other words do they provide incentives to improve fit and interplay given certain constraints?

The incentive perspective, as proposed by Kaufmann-Hayoz and Gutscher (2001), has the advantage of taking the rationale (i.e. functioning mechanism), the actors involved in implementation, the target groups and the implementation and enforcement requirements of policy instruments into account. In an ideal case, from a policy implementation point of view, the behavior of targeted groups or individual actors is adjusted in such a way that natural resources and the ecosystem services they provide are used, i.e. conserved, in socially optimal ways (Baumol and Oates, 1988). In practice, the optimal level is generally not known. However, stakeholders do have preferences regarding environmental management and ecosystem service provision which are usually not attained through markets alone. Since IWRM implementation is essentially based on existing environmental policies it has been proven useful to build on these as much as possible and to fill the gaps through complementary tools forming part of a general policy mix (cf., Ring and Schlaack, 2011; Grunewald and Bastian, 2012). This is especially important in the context of IWRM implementation as it involves adjustment of different existing policy instruments in many different policy sectors. Thus, different existing policies have to be taken into account in order to achieve the objectives of IWRM. Besides creating a new spatial planning unit, the river basin, IWRM implementation requires new forms of cross-sectoral cooperation and cross-jurisdictional collaboration. At best, all relevant stakeholders are involved in this process and participation is encouraged.

Policy instruments can be classified according to their major characteristics. Different authors and disciplines have suggested different categorizations, but the following three are widely used in the literature (Michaelis, 1996; Gunningham and Young, 1997; Sterner, 2003):

- Regulatory, command-and-control instruments, including permits, standard-setting and land use zoning or planning, directly control or restrict environmentally damaging activities.
- Economic instruments, such as environmental taxes, charges and fees, put a price on environmentally damaging behavior, thus internalizing negative externalities, whereas payments for environmental services and ecological fiscal transfers reward conservation enhancing behavior, thereby addressing positive externalities.
- Communication (information) and diffusion instruments aim to shift individual or community preference functions towards more conservation and inform or educate people about relationships between their activities and the environment (Ring and Schlaack, 2011).

Table 3.10 provides an overview of these policy instrument categories, including the instrument of collaborative agreements, with regard to their rationale, actors involved, target groups and implementation requirements.

In practice, these instruments are often used in combination. In some cases, instruments have been intentionally introduced to enhance the outcome of others.

While in the past traditional water management was principally based on sectoral command-and-control regulations and land use planning according to administrative boundaries, the IWRM approach requires additional measures to solve the resulting problems of fit and interplay when a new basis for land use planning is introduced with the river basin approach. Gunningham et al. (1998) consider most existing approaches to regulation as seriously sub-optimal, meaning "[...] that they are not effective in delivering their purported policy goals, or efficient, in doing so at least cost, nor do they perform well in terms of other criteria such as equity or political acceptability". The suitability of policy instruments is likely to be context-specific, depending highly on the characteristics of the environmental issue under consideration (Opschoor and Turner, 1994; Gunningham et al., 1998). Thus, effective policy instruments addressing point-source pollution from industries are likely to be very different from those to avoid land degradation. Besides different environmental problem characteristics, institutional actors, the political and economic contexts in which policy mixes must be designed differ as well. Hence, there is no single optimal policy instrument applicable to all circumstances.

Zysset and Kempter (2013) studied IWRM implementation in Switzerland on the regional and local level. The authors confirm that the objectives of IWRM require the application of many different policy instruments. While each policy instrument has specific advantages and disadvantages, synergies and incompatibilities, Zysset and Kempter (2013) could not identify a specific instrument category that is especially suited for IWRM implementation. Communication instruments such as participatory processes and influencing of values, norms, knowledge and ability, for instance, can help transferring the rationale for an integrated approach, however, they do not bring about the balancing between the often conflicting water uses (Zysset and Kempter, 2013). They should rather be combined with command and control instruments or collaborative agreements to contain the system drivers and to give some latitude to its most dependent participants. Hence, reliance on a variety of different policy instrument types is useful since there is no single best instrument category to foster integration and adaptive capacity.

Instrument type Rationale	Implementing actor(s)	Target group(s)	Implementation and enforcement
Command and control instruments			
Legal prescriptions having a direct impact on the range of options open to specified actors, constraining certain ways of acting or excluding some forms of conduct.	Only legislative public authorities (federal, state or municipal for instance) are legitimized.	Apply to all actors as defined in the legal norm in the same way.	Enforcement mechanisms necessary and heavily demanding on technical, financial and personnel resources; granting of exceptions in practice recurrently compromises efficiency; prescriptions towards barely accessible actors are difficult to enforce and control.
Economic instruments			
Internalization of externalities of economic activities through price signals: raising the cost of polluting behavior, reducing the cost of or rewarding for environmentally sound behavior and establishing markets for polluting rights or positive externalities	Directly through public authorities if legal basis is required or on a voluntary basis through any stakeholder (public authorities indirectly involved).	Direct target groups are public and private consumers, indirectly all stakeholders.	Legal basis usually required. Simple and flexible rules facilitate implementation and reduce room for dispute or manipulation. Monitoring and enforcement is needed.
Collaborative agreements			
Legally binding or non-binding agreements of private stakeholders towards other stakeholders, esp. public authorities, based on negotiations to enhance policy goals.	Actors involved in the agreement mutually develop and implement the instrument.	Target group (agreement partners) is involved in development and implementation.	Although intended to be voluntary, participation is often enforced under slight pressure from public authorities.
Communication and diffusion instruments			
Aim at influencing actors' goals (preferences), knowledge and behavior by modifying motivational, cognitive, and social preconditions of action through stimulating the thinking of individuals and shaping societal discourses on goals / options for action.	Available to and implementable by all actors.	General public or specific groups of any kind.	Suitable for use in preparation of other interventions or as a complement. Instrument often a precondition for proper functioning of other instruments.

Table 3.10: Instrument typology with regard to actors involved, target groups and implementation requirements; based on Kaufmann-Hayoz and Gutscher (2001)

Typically, regulatory, planning, policy-making and enforcement powers are divided between water authorities representing federal, state or municipal governments. Their responsibilities are often strongly segregated between different sectors of public policy to be controlled into separable individual elements. Moreover, environmental management has relied heavily on regulatory instruments in the past, often without any participatory, cooperative forms of governance beyond formal consultation exercises (Moss, 2004b). Thus, control is exerted centrally, adhering to rigid and detailed plans for the fulfillment of established goals. This predominance of 'command-and-control' instruments and technocratic forms of policy-making has been widely attributed as cause for the low level of environmental policy implementation in the past.

Besides the different ways how policy instruments influence actor's behavior, they also have specific advantages and disadvantages as summarized in Table 3.11. For instance, to address point-source problems of resource use (water resources used as a pollution sink or as a source) command and control measures and also economic instruments have been successful. These problems are not (primarily) political boundaries crossing problems and can be addressed according to existing jurisdictions. To control diffuse problems other instruments have been used with more success.

Command and control instruments are regulatory instruments that are direct and mandatory, representing legal prescriptions with a direct impact on the options open to specified social actors. Hence, command and control instruments constrain certain ways of acting or exclude some forms of conduct. Mandatory orders are used to prescribe or prohibit specific actions and/or specific outcomes of actions stating clearly what is legal and what is illegal. The rationale of

Advantages	Disadvantages
Command and control instruments (such as emission limitation, planning regulations, spatial zoning)	
• Verifiable, reliable and predictable in their main impact if they are enforced • Allow positive economies of scale when widely-used	• Require precise knowledge of activities, dependencies / options of those involved; complex to develop • Inflexible, possibly inefficient or involving side effects • Can be complicated to verify • Can be resisted / disregarded if benefit is unclear • Do not motivate to exceed the minimum standards
Economic and financial instruments (such as subsidies, polluter taxes, auctioning)	
• Can promote an economic approach if associated with privatization of public goods • Can create incentives to exceed minimum standards • Can reduce enforcement costs for the authorities • Allow cost-efficient solutions in market situations	• Impact is hard to predict as action is transferred to market players • Taxes and grants can stifle innovation • Can generate high subsidy costs in some circumstances • Can lead to unfairness towards non-beneficiaries
Collaborative agreements (such as public private partnerships, certification and labels)	
• Can be very efficient and effective if the interests of the participants are at least partially parallel • Allow mutual motivation / control among participants • Flexible and practical	• Can lead to unclear roles of public and private participants • Complicated to enforce • Can stifle competition and exclude third parties • Sanction options are often limited
Communication instruments (such as influencing of values, norms, knowledge and ability, participative processes)	
• Can extend the number of participants • Rapidly implemented, can be motivational • Can supplement other policy instruments well	• Their impact is uncertain and hard to control • Can be complex, slow and short-lived • Not appropriate for strongly conflicting interests

Table 3.11: Categorization of policy instruments and their advantages and disadvantages; based on Gunningham et al. (1998); Kaufmann-Hayoz and Gutscher (2001); Australian Government (2009) from Zysset and Kempter (2013)

command and control instruments is based exclusively on command, control, and sanction, thus, it is assumed that in order to avoid sanctions actors behave according to the prescription or norm (Kaufmann-Hayoz and Gutscher, 2001).

Penalties and other enforcement mechanisms are essential for the implementation and effectiveness of command and control instruments. Measures of enforcement can be based on a variety of (conditional or secondary) instruments, ranging from withdrawal of use permits to criminal prosecution, and often include an economic component in addition to the pure regulatory aspect (e.g. fines if emission levels are exceeded).

In most democracies, only the legislative authorities have the legitimization to use command and control instruments. Usually, this legitimization is shared by different administrative levels (e.g. federal, state and municipal governments) depending on the relevant policy domain. The principal application and eventual enforcement measures of command and control instruments is in the hands of the public authorities. Thus, policy design and application is, in practice, generally limited to public actors.

Since command and control instruments apply to any actor or group of actors specified in the legal norm in the same way, the instrument can be used to influence the behavior of any target group, from individuals to corporate actors or private companies.

Implementation and enforcement of command and control instruments often places heavy demands on technical competence as well as on the amount of available human and financial resources (Mayntz, 1980) and can be troublesome in many respects (Kaufmann-Hayoz and Gutscher, 2001). In practice, a common phenomenon is that granting of exceptions (legally or illegally) often compromises the implementation of efficient command and control instruments. These exceptions are rarely appealed against by other public authorities. Although in principal any target group can be addressed, command and control instruments directed to barely accessible target groups are often very difficult to enforce and hardly controlled. In this case the fear of sanctions for illegal behavior may disappear.

Direct regulation or command and control regulations have been used in water management in areas where there is a high risk which could result in a substantial impact on the economy, environment, groups or individuals. The protection of water source and nature conservation areas or standards for fertilizer and pesticide uses in agriculture are typical examples. In case where there have been problems with systematic noncompliance with self-regulation or when the risks of non-compliance are high to society, these instruments should be considered. Command and control instruments work best when actors are easily identifiable and accessible.

Traditional command and control instruments use a compulsory application approach where it is decided directly (sometimes unilaterally) what situation is desired, e.g. for the water body, while using the power of the state to achieve it. Therefore, Porto and Lobato (2004) argue that from the perspective of an omnipresent state, this appears to be enough to achieve the intended objectives, but “effective implementation presents deficiencies that result from the fact that the quality of the environment in general and of the water resources in particular, is the result of the action of multiple social agents. This makes it rather complex to “command” all the factors involved to achieve the desired objectives, including those to impose law-enforcement mechanisms which require structures to inspect and apply fines and penalties, with increasing difficulties because of the magnitude of the problem”.

A general disadvantage of command and control instruments is their rigidity, because some generalizations have to be made to set regulatory rules and these rules apply to the entire target group equally, which may hinder the efficiency and effectiveness of environmental policies (Kaufmann-Hayoz and Gutscher, 2001). These may not take into account the differences in control costs among the polluting agents or those who exploit the natural resources. Since rules of command and control instruments, e.g. land use standards, are generically imposed, they do not provide to those holding advantages in reducing their externalities to levels lower than the others (cf., Porto and Lobato, 2004). This can result in discarding of more efficient alternatives to achieve environmental policy goals.

In the 1960s public authorities in developed countries started applying command and control institutions, often as a combination of ambient environmental quality standards, technology requirements, emissions criteria, and other restrictions to control use of natural resources (John, 1994). According to Lubell et al. (2002), most commentators agree that these instruments have successfully reduced pollution from well-defined point sources like factories and sewage treatment plants. The use of these instruments for regulating geographically diffuse non-point sources of pollution emitted by or resulting from individual actions of numerous and heterogeneous resource users who jointly affect the environmental quality of a watershed, has been much less successful (John, 1994; Lubell et al., 2002). Moreover, command and control instruments have difficulty in addressing environmental issues that involve different media (e.g. air, water and soil) and span political and administrative boundaries (John, 1994; Marsh and Lallas., 1995; Lubell et al., 2002). Moreover, in applying command and control regulations it is supposed that actors have not violated commanded rules until a violation of rules is proven or demonstrable violations can be confirmed. Thus, especially in case of diffuse, continuous and poorly understood effects of human action the use of these instruments presents several intrinsic weaknesses.

Besides the disadvantage of command and control instruments being less suited for environmental problems of diffuse character, they rely on a strong role of the state in designing, implementing and enforcing the instrument. The experience with command and control instruments in developing countries has been disappointing so far. Specific problems of application of these instruments in developing countries: definition of standards (e.g. for land uses that conflict with poor smallholder farmers), control and enforcement is often difficult or impossible because of a lack of resources, penalties on poor farmers prove ineffective, presence of corruption and weak governmental and legal systems. In the context of a weak state, as is often the case in developing countries, it is reasonable for the state to concentrate on core activities and to look for opportunities of cooperation to achieve other environmental goals. If command and control instruments are complemented by policy instruments that reduce the role of government while integrating other actors in policy design and implementation, the use of incentives to attract additional actors to participate becomes apparent. Hence, other instruments such as economic instruments, communication and information tools and collaborative agreements present opportunities for others to become part of the policy process.

However, in order to achieve more context-specifity it is necessary to add complementary policy instruments to regulatory command and control instruments. Other instruments, especially economic instruments, are more flexible in achieving different degrees of environmental targets (beyond compliance), involving additional actors for instrument design (information gap) and implementation, addressing specific local contexts (administrative gap) and in tapping additional resources (funding gap, information gap).

The principle rationale for the use of classical economic instruments is the assumption that environmental degradation and resource depletion occur because a substantial part of the costs or benefits of economic activities, so-called externalities, is not being paid by the actors responsible but by the general public, e.g. in the form of environmental damage, security and health risks, or long-term climatic risks (Kaufmann-Hayoz and Gutscher, 2001). There are three principal forms of intervention conceivable:

- Raising the costs of polluting behavior based on the principle of true costs. This approach imposes taxes or charges for environmentally harmful behavior to cover the costs of the damage it causes. This way the actor’s range of options is not influenced directly insofar as they are not constrained by regulations. However, the imposition of economic costs for environmentally harmful behavior is an attempt to steer actor’s behavior toward more environmentally sound actions, because these are more cost-efficient. In a functioning market mechanism, the least expensive actions are supposedly to be realized first.

- Reducing the costs of environmentally sound behavior is an attempt, contrary to charging environmentally harmful behavior, to reward environmentally sound behavior, often by the means of subsidies or lower prices for environmentally responsible behavior. These positive incentives aim at making environmentally sound behavior more attractive, thus, more likely.
- Establishing markets for polluting rights for the trading of resource use permits, e.g. to pollute the air. The use of environmental resources as pollution sink requires a permit which has to be bought and can be traded.

These classical forms of economic instruments are usually implemented by public authorities on a legal basis at different political levels (municipal, state, international etc.), depending on the authority responsible for the traded good and the nature of the good. However, economic instruments can also be implemented by other actors than public ones as long as this is permitted by the existing legal basis. The target groups of economic instruments implemented by public authorities are generally private actors and households as well as companies. Although being targeted directly to specific actors and actor groups, economic instruments usually indirectly affect all actors through price effects (Kaufmann-Hayoz and Gutscher, 2001). Economic instruments affect incomes of different actors through the modification of prices as well as through the way how tax revenues are used, i.e. subsidies financed.

The implementation of economic instruments requires a legal basis. Moreover, it has to be clear to whom, for what purpose and how the economic instrument is to be applied. The latter requires a definition of how the charge, reward or specific action has to be calculated, i.e. economically valued and how revenues are to be spent. This is done on the basis of the law or specific contractual agreements. Simple and flexible regulations based on uniform calculations with little room for administrative interpretation are particularly important for successful implementation and less vulnerable to manipulation and dispute (Kaufmann-Hayoz and Gutscher, 2001). Just as for any command and control instrument, enforcement and monitoring is needed. The effort involved in enforcement and monitoring can vary widely. For political acceptance of the instruments it is crucial how the revenue is spent or how subsidies are financed. The use of revenues to support targeted environmental goals or redistribution to the public will often increase acceptance (OECD, 1997). Economic instruments in water management are often used to distribute benefits and costs resulting from land use impacts on water resources, including direct or indirect subsidies, taxes, and transferable property or use rights for land, water, and emissions (Amezaga, 2006).

Command and control as well as traditional economic instruments are principally aimed at modifying different aspects of the external structure of actor's behavior: the political, legal, and administrative, the socio-economic, and the physical environment (see Figure 3.9). Most command and control instruments directly restrict actors' options, while economic instruments attempt to make environmentally desirable actions more attractive or rewarding and undesirable actions less attractive by providing appropriate incentives. However, actor's internal structures are generally influenced more by communication and diffusion instruments. These instruments address the knowledge and goals of individuals and organizations and attempt to influence their respective ways of achieving them as well as affecting their perceptions and appraisals of the social and physical reality (Kaufmann-Hayoz and Gutscher, 2001).

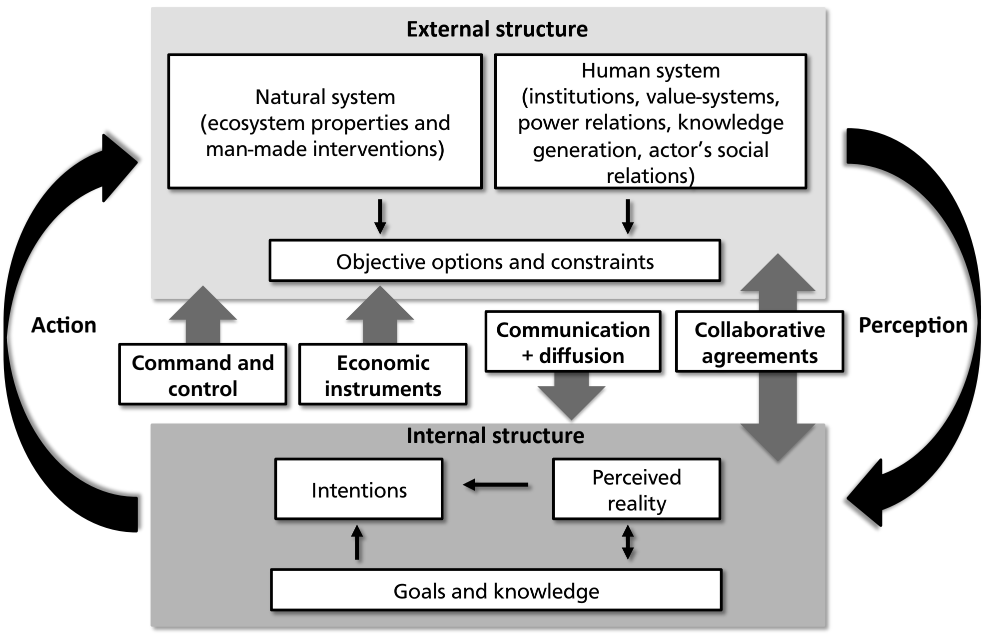

Figure 3.9: Primary target of different types of instruments; based on Kaufmann-Hayoz and Gutscher (2001)

Policy challenge	Description
Heterogeneity and multiple objectives	IWRM requires cross-sectoral integration considering all water and land uses affecting the resource
Irreversibility	Beyond certain thresholds or tipping points, impacts may be irreversible and cause ecosystem collapse; appropriate policies adhere to the precautionary principle
Information gaps	The inherent complexity of ecosystems to be managed requires policy decisions under uncertainty; adaptive management approaches
Mix of values	River basins provide a variety of use values and non-use values, some of which are tangible and marketable, whereas others are of a public or common good nature
Multiple market failures	Both negative and positive externalities need to be addressed through economic instruments and regulations
Mix of pressures	Different pressures on river basins arise from various economic sectors, calling for different responses
Impact accumulation	Small impacts over a long time may create large losses with irreversible outcomes in the long-run, while the costs of prevention have to be incurred in the present
Spatial externalities	Benefits and cost are often incurred at different places. Costs are also unequally distributed between economic sectors, and unevenly spread across administrative units
Multi-level governance	IWRM policies require appropriate instruments at local, regional, national and international level
Multi-actor governance	Due to the multi-faceted nature of IWRM, both public and private actors need to be involved, next to the increasing relevance of hybrid organizations crossing the public-private divide

Table 3.12: Policy challenges of IWRM implementation calling for a policy mix; based on (Ring and Schlaack, 2011)

Collaborative agreements between individual actors and organizations often target problems beyond the scope of existing regulations. These instruments aim at the internal structure of actor's, through increasing their knowledge and motivation for potential gains from collaboration, as we as at their external structure in making agreed actions more attractive. The enlargement of the set of relevant actors within the river basin boundary involved in the policy-making process can lead to new forms of collaboration. Lubell et al. (2002) stresses that so-called *watershed partnerships* can build on local knowledge and craft specialized policies congruent with local watershed problems. Moreover, the authors stress that voluntary participation by local actors allows for the development of self-monitored norms of cooperation that circumvent costly legal and administrative compliance mechanisms as they would be required for command and control instruments. These watershed partnerships emerge because they produce mutually beneficial solutions to resource conflicts in the watershed that are (Pareto) superior to command-and-control institutions (Lubell et al., 2002). However, actors often need to perceive an incentive for collaboration in order to invest in social efforts. Lubell et al. (2002) argues additionally, that the greater the transaction costs of developing and maintaining partnerships, the less likely partnerships will emerge.

Operational constraints caused by resource scarcity require instruments that incorporate additional resources apart from governmental ones. Thus, instruments seem promising that are flexible in addressing different contexts, involve non-governmental stakeholders in resource demanding tasks (e.g. participation in land use planning, decision making processes) and possibly tap additional funds. Positive experience has been made with voluntarism in Australia with its Landcare movement, which aims at improving natural resource management by landholders and community groups (Australian Government, 2009). Voluntarism is often initiated by a governmental, or in developing countries often a non-governmental, intermediary playing the role of coordinator or facilitator, providing administrative support and targeted funding. However, participation is completely voluntary. This approach has proven most successful where individuals or businesses perceive their self-interest to be consistent with the broader public interest or at least other involved parties' interest. Although it is often impossible to involve all stakeholders, a voluntary approach "can be a very useful first step where achieving a threshold of cultural change is required before other policy instruments can be contemplated, and/or where active participation by a large number of individuals is required to solve a problem (such as reducing water usage or the adoption of better natural resource management techniques)" (Australian Government, 2009). An important advantage of voluntarism is its non-interventionist character implying low (governmental) resource requirements and high political acceptability by involved stakeholders because of low degrees of prescription and coercion by the instrument (cf. Gunningham et al., 1998). At best, voluntarism results in stakeholders internalizing the motivation for the desired behavior and achieving sustainable, long-lasting behavioral change. However, there are significant possible disadvantages as well. Once again, effectiveness depends strongly on the motivation to participate and it may be difficult to target and monitor outcomes without incurring high administration costs (Australian Government, 2009).

Given the diversity of policy instruments, it is important to note that combinations of different policy instruments work best since each instrument category has something valuable to offer while at the same time having substantial limitations as a stand-alone strategy for government intervention. The advantages of dependability and predictability of properly monitored and enforced command and control regulation, for example, are contrasted with its properties of being inflexible, time consuming to make and amend, costly to enforce and inefficient when not enforced properly. Economic instruments, instead, are often efficient but their actual impact can be uncertain. Communication and collaboration instruments are less coercive and intrusive than command and control regulation and generally cost-effective, but they tend to be unreliable when used in isolation (Australian Government, 2009). Hence, a combination of instruments instead of relying on a single type can be the best way of overcoming the deficiencies of individual instruments, while taking advantage of their strengths. Table 3.12 summarizes major policy challenges of IWRM implementation which, according to Ring and Schlaack (2011), need to be addressed by a policy mix rather than individual policy instruments alone.

In general, a top-down approach encounters difficulty in being sufficiently context-specific with regard to specific natural or human system contexts. This has resulted in problems of 'full' integration only for the sake of integration. To encounter the right context, thus, requires a certain degree of flexibility in management design that cannot be provided equally by all management instruments. Moreover, when it comes to cross-sectoral and cross-administrative cooperation and collaboration this can hardly be achieved by mandate but rather through specific incentives and active involvement of stakeholders in the IWRM implementation process. Management instruments have to provide appropriate incentives for cooperation and collaboration, i.e. interaction among relevant stakeholders for a given management context. Furthermore, problems of top-down approaches relate to the capacity to implement, monitor and enforce management instruments and specific measures. Local stakeholder or target groups are often much more capable out of executing these necessary tasks than higher order administrative bodies. The requirements of complementary management instruments can be summarized as follows:

- Flexible enough to match human and natural system context - address ecosystem characteristics and interactions of humans with the natural system at the right scale.
- Provide incentives for cooperation and collaboration across sectors and administrative boundaries at the identified scale.
- Encourage social learning, self-regulation and participation.
- Management instruments have to be embedded in a diverse institutional and social system (compatibility with existing regulations)

This variety of requirements calls for properties of different instruments to be applied in a policy mix. More flexible solutions to meet specific context can be achieved through economic instruments as well as through collaborative agreements, both bear the potential to reduce expenses of public authorities and to involve additional stakeholders in the instrument design and implementation process. Communication and diffusion may also play an important role in pronouncing new ways of policy and decision making in the context of IWRM. By applying additional policy instruments to traditional command and control instruments the decision making level may possibly be closer to the actors involved. The actors in turn may have more incentives for cooperation and coordination. This desired effect is illustrated in Figure 3.10.

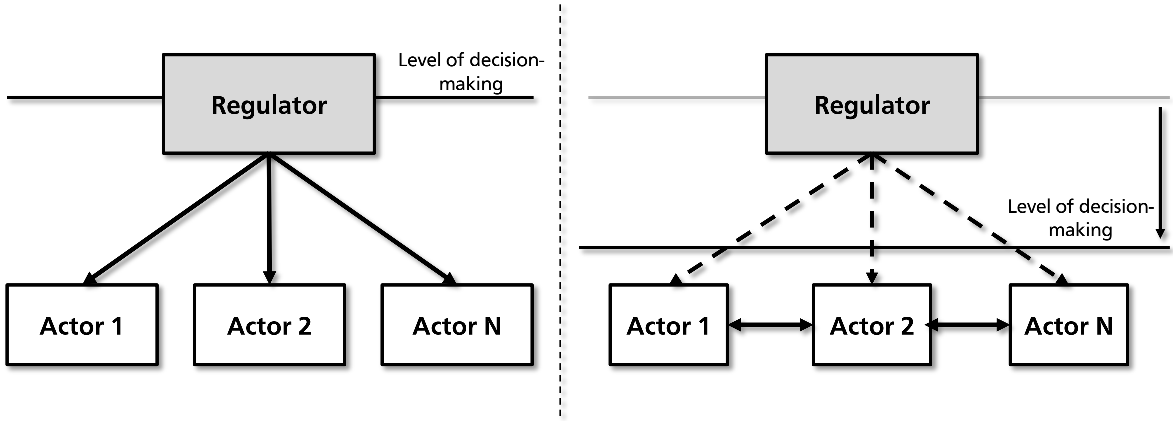

Figure 3.10: Interaction between regulator and actors. On the left based on top-down command and control regulation; on the right additional policy instruments move decision-making level closer to addressed actors and provide incentives for cooperation (Author's work)

Payments for Ecosystem Services (PES) represent a promising instrument that combines several of the desired instrument properties. Over the last two decades, PES has become a popular instrument to complement protected areas and other regulatory approaches in ecosystem conservation policy. According to Ferraro (2011), the popularity of PES schemes in developing countries resulted from frustration and ethical issues associated with regulatory approaches and from dissatisfaction and criticism of indirect approaches. Since the introduction of PES schemes is accompanied by the hope to raise additional funding, concerns over inadequate conservation budgets have also played an important role. Moreover, Maro (2010) argues that "unlike conventional approaches to conservation which are regulatory and top-down, PES is incentive-oriented and provides for free interaction among the different parties".

With the introduction of Payments for Hydrological Ecosystem Services (PHES), the instrument is also applied specifically to environmental challenges of water resources management. PHES combine governance and management, recognizing their interdependence. The ecosystem service concept as governance system forms the basis to enable PHES as a practical management instrument.

Although principally an economic instrument, PHES contain several characteristics of collaborative agreements as well as communication and diffusion instruments. The voluntary nature of the instruments is a common feature of collaborative agreements and the ecosystem service approach as the conceptual basis of PHES is a strong communication instrument.

The potential role of PHES in IWRM implementation will be assessed in the following chapters.

3.5 Summary

In order to improve the operationalization of IWRM the principal problems of fit and interplay have to be addressed. In general, attempts of IWRM implementation apply command and control approaches to achieve institutional fit at the river basin level. The resulting problems of institutional interplay are addressed through political or membership linkages on the basis of the institutional solution of fit. The interdependence of fit and interplay is not recognized and the problems are solved in sequence, in practice solutions to interplay follow solutions to fit, rather than in interaction. This may result in sub-optimal scales for management, low cross-sectoral stakeholder involvement and cross-jurisdictional collaboration. Although the majority of operationalization approaches considers a policy mix, e.g. command and control instruments complemented by communication instruments, the problem of finding the right management scale with context-specific solutions to fit and interplay is not explicitly addressed. The 'standard packages' of IWRM implementation in developing countries are principally based on the role of the state as designer and implementer as well as enforcement actor. Given the prevailing resource constraints in developing countries, this does not seem to be the best solution. Policy instruments that are less interventionist, i.e. concentrating state resources for core tasks, and building on the involvement of additional actors for the design, implementation and enforcement of policy actions are a promising complement to existing policies.

Economic instruments or incentive-based instruments are supposed to be more flexible and efficient in achieving environmental policy goals. Toward the end of the 1990s, economic instruments where introduced in developing countries, especially in Latin America, for the conservation of protected areas and to combat deforestation in general. The instrument of Payments for Ecosystem Services (PES) has gained broad popularity among the variety of economic instruments and is now applied more or less in all Latin American countries and several other developing countries in Africa and Asia. In addition, conservation efforts for protected areas are now widely applied, where the protection of water resources taking the form of Payments for Hydrological Ecosystem Services (PHES) is concerned. These instruments are praised by practitioners and scientist as promising and innovative tools to address problems of environmental degradation. Their application in the context of IWRM has received little attention so far. However, their popularity and widespread application in developing countries as well as their proximity to the goals of IWRM gives reason to analyze their suitability as a possible alternative or complement to existing approaches to IWRM operationalization. PHES are based on the concept of Ecosystem Services (ES) for environmental governance. The ES concept departs from quite a different point than the traditional government perspective of environmental regulation. Instead of political and membership linkages it focuses on functional linkages between the natural and the human system. Thus, it provides a different way of addressing fit and interplay. The following chapter introduces the ES concept and analyzes, based on the findings of Sections 3.1 and 3.2 of this chapter, its suitability to address the problems of institutional fit and interplay in order to improve operationalization of IWRM through instruments based on its principals such as PHES.

4 The concept of Ecosystem Services and their valuation

The present and the following chapter provide a general theoretical analysis of the potential of Payments for Hydrological Ecosystem Services (PHES) to contribute to improvements in IWRM implementation and operationalization as a possible solution statement (see Figure 4.1). The Ecosystem Services (ES) concept, as the basis of PHES, is addressed in this chapter. The principal aim is to analyze whether the concept provides a means to address the governance challenges of fit and interplay in the context of IWRM implementation. Moreover, it will be analyzed if the concept of hydrological ecosystem services can contribute to identifying the appropriate scale for management, hence, guiding operationalization of IWRM for specific contexts.

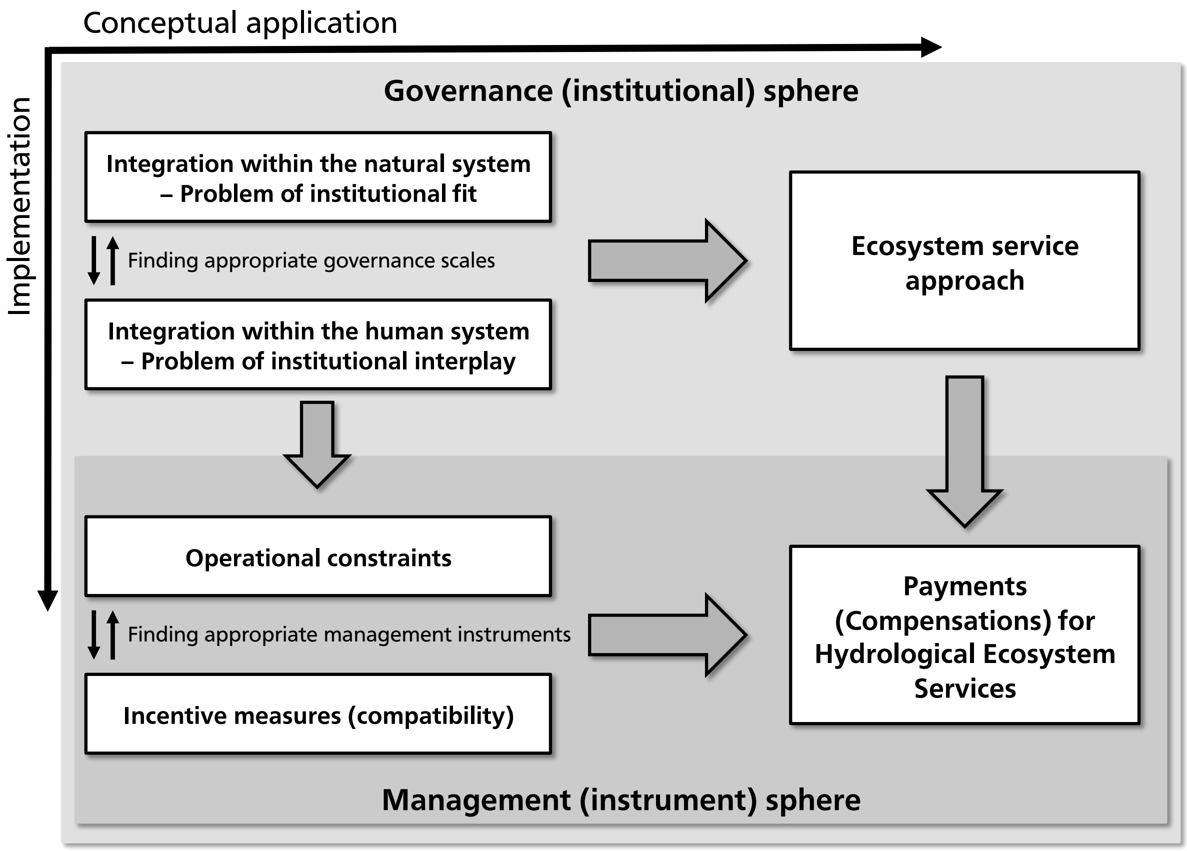

Figure 4.1: Illustration of the methodological conceptualization of core IWRM implementation problems and the PHES solution statement based on the ecosystem service concept (Author's work)

4.1 Introduction to the ecosystem service concept

Basic ecological concepts represent the main point of departure for the concept of Ecosystem Services (ES). Ecosystems are understood as dynamic complexes of living biotic communities of plants, animals and microorganisms which together with their non-living inorganic environment form a functional unit with the capacity of self-regulation to a certain degree (cf. Tansley, 1935; UN, 1992; MEA, 2005). For practical reasons it is often necessary to define the spatial extension of an ecosystem of concern, e.g. a lake or a river basin, although this is merely conceptual and not based on any distinct spatial configuration of interactions (Tansley, 1935; Odum, 1969). However, ecosystems may also nest into each other. Apart from a specific species composition, an ecosystem also comprises interactions and processes within a certain habitat including all interdependencies within all of its components. This encompasses further all ecological functions, e.g. regulation of nutrients and climate, resulting from the respective energy and material flows between different compartments. Ecosystem functions are a subset of the interactions between the structure and processes of an ecosystem that underpin the capacity of an ecosystem to provide goods and services (TEEB, 2010a). The structure of an ecosystem refers to its biophysical architecture made up of a varying composition of species. Ecosystem processes, however, describe any change or reaction (physical, chemical or biological) occurring in an ecosystem including decomposition, production and fluxes of energy and nutrients. Table 4.1 illustrates examples of ecosystem processes that comprise ecosystem functions important for the provision of ecosystem services.

Ecosystem function		Exemplary ecosystem processes
Primary production	⟷	Photosynthesis, plant nutrient uptake
Decomposition	⟷	Microbial respiration, soil and sediment food web dynamics
Nitrogen cycling	⟷	Nitrification, denitrification, nitrogen fixation
Hydrological cycle	⟷	Plant transpiration, root activity
Soil formation	⟷	Mineral weathering, soil bioturbation, vegetation succession
Biological control	⟷	Predator-prey interactions

Table 4.1: Examples of ecosystem processes that comprise ecosystem functions important for the provision of ecosystem services; based on (Virginia and Wall, 2000) as cited in (TEEB, 2010a)

An important term with regard to ecosystems and the provision of services is the term biodiversity or biological diversity. Biodiversity describes the variability among living organisms from all sources (terrestrial, marine, aquatic ecosystems etc.) and the ecological complexes of which they form part of. This includes diversity within species, between species and between ecosystems (MEA, 2005). Moreover, the term biodiversity reflects the hierarchy and complexity at the level of genes, individuals, populations, species, communities, ecosystems and biomes (TEEB, 2010a). Changes in ecosystem functioning can be a result of alternations in biodiversity. Hence, biodiversity at all levels contributes to the maintenance of ecosystem functions and services (Hanna et al., 1996). Several linkages between changes in biodiversity and the way ecosystems function have been identified in theoretical and empirical studies (Schulze and Mooney, 1993; Loreau et al., 2002).

Generally, ecosystem functions are referred to as Ecosystem Services if an anthropogenic benefit can be derived from ecosystem functioning (Loft and Lux, 2010). Barbier et al. (1994) defines ecosystem services more precisely as those ecosystem functions that are currently perceived to support or protect human activities or affect human well-being (cf. Hanna et al., 1996). Natural resources, i.e. ecosystem goods, as well as ecosystem services are generated and sustained by ecosystems. Moreover, they also maintain nature in a condition attractive to humans (Hanna et al., 1996). One of the first mentions of Ecosystem Services was by Ehrlich and Ehrlich (1981) and (Ehrlich and Mooney, 1983). De Groot (1992) and Daily (1997) produced a further development of the ES concept. Since then there have been many attempts to define the term Ecosystem Services, whereas the following definitions remain very influential and popular:

- “The conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life” (Daily, 1997)
- “The benefits human populations derive, directly or indirectly, from ecosystem functions” (Costanza et al., 1997)
- “The benefits people obtain from ecosystems” (MEA, 2003)

The definition of the Millennium Ecosystem Assessment (MEA), listed last in the above, has its origin in the former two definitions, following Costanza et al. (1997) in including both natural and human-modified ecosystems as sources of ecosystem services, while adapting the definition of Daily (1997) in using the term “services” to encompass both the tangible and the intangible benefits humans obtain from ecosystems, which are sometimes separated into “goods” and “services” respectively.

While many authors (Daily, 1997; Costanza et al., 1997; MEA, 2005; de Groot et al., 2010) do not distinguish ecosystem services from ecosystem goods, several other authors do this explicitly (Boyd and Banzhaf, 2007; Wallace, 2008; Fisher and Turner, 2008; Balmford et al., 2008; Bateman et al., 2011). Assessments of connections between people and nature are complex and different disciplines look at them in different ways which makes it very challenging to find common categories and classification systems. Additionally, the objectives for such assessments differ, e.g. the need to describe ecosystem services in order to map or value them economically or to estimate the human impact on ecosystems and how this impact changes their capacity to deliver services with the aim to develop appropriate policies (Haines-Young and Potschin, 2013). Hence, the need for a common accounting system for natural capital has led to the Common International Standard for Ecosystem Services (CICES) as a contribution of the European Environment Agency (EEA) to the United Nations Statistical Division (UNSD) as part of the revision of the System of Environmental-Economic Accounting (SEEA). CICES intends “to help negotiate the different perspectives that have evolved around the ecosystem service concept and assist in the exchange of information about them” (Haines-Young and Potschin, 2013). The widely employed and acknowledged classification of ecosystem services used in the Millennium Ecosystem Assessment (MEA, 2005) incorporated four ecosystem service categories: provisioning, regulating, cultural and underlying supporting services (see Figure 4.2).

While the MEA had its main focus on the assessment of state of the earth’s ecosystems, communicating the intrinsic dependence of human well-being on intact ecosystems and at the same time the destruction of those by human activity, an

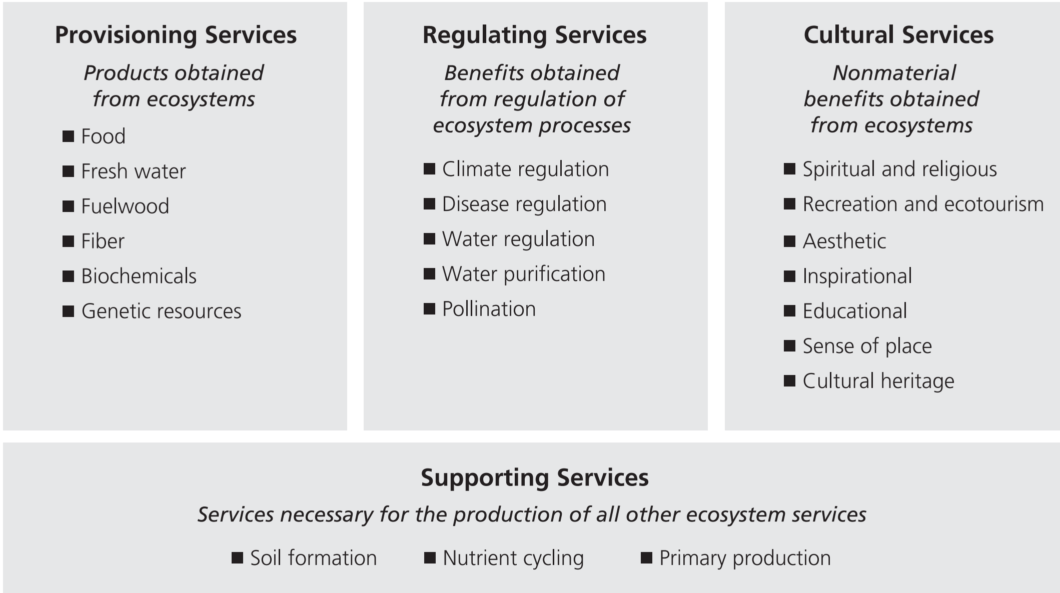

Figure 4.2: Overview of ecosystem service categories and exemplary ecosystem services (MEA, 2005)

international study The Economics of Ecosystems and Biodiversity (TEEB) launched in 2007, aims to draw attention to the global economic benefits of biodiversity and ecosystem services. The TEEB study, in its ecosystem service typology, basically follows the categorization of the MEA by using the familiar provisioning, regulating and cultural (and amenity) categories, but omits the category of supporting services and introduces *habitat services* as a new one. The omission of the supporting services category is justified with a clearer conceptual distinction of ecosystem services and underlying ecosystem functions and processes which, it is argued, many of the supporting services (e.g. nutrient cycling or food chain dynamics) represent (TEEB, 2010a). The *habitat service* type is introduced to put explicit emphasis on the importance of ecosystems to provide habitat for migratory species and gene pool protectors in the context of biodiversity. Several other typologies have also been debated among the scientific community (Wallace, 2008; Fisher and Turner, 2008; Costanza, 2008) demonstrating a lively debate on how to tackle human and nature relationships. This underlines the rationale for the CICES initiative.

CICES Theme	CICES Class	Corresponding TEEB Categories			
Provisioning	Nutrition	Food	Water		
	Materials	Raw Materials	Genetic resources	Medicinal resources	Ornamental resources
	Energy				
Regulating	Regulation of wastes	Air purification	Waste treatment		
	Flow regulation	Disturbance, prevention or moderation	Regulation of water flows	Erosion prevention	
	Regulation of physical environment	Climate regulation	Maintaining soil fertility		
	Regulation of biotic environment	Gene pool protection	Life cycle maintenance	Pollination	Biological control
Cultural	Symbolic	Information for cognitive development			
	Intellectual and Experiential	Aesthetic information	Inspiration for culture, art and design	Spiritual experience	Recreation & tourism

Table 4.2: CICES classification of ecosystem services and respective TEEB categories (Haines-Young and Potschin, 2013)

In a very first step, ecosystem services are defined for CICES as the contributions that ecosystems make to human well-being, arising from the interaction of biotic and abiotic processes, and refer specifically to the *final* outputs or products from ecological systems (Haines-Young and Potschin, 2013). Thus, ecosystem services are understood as things directly consumed or used by people. The TEEB typology, the classification which is currently elaborated for CICES, recognizes provisioning, regulating and cultural services as final ecosystem outputs but it does not cover supporting services as originally defined in the MEA.

Table 4.2 illustrates how the CICES classification of ecosystem services relates to the categories used in the TEEB study. Accordingly, since supporting services are only indirectly consumed or used they are considered as part of the underlying structures, processes and functions that characterize ecosystems (Haines-Young and Potschin, 2013). Additionally, supporting services often simultaneously facilitate the output of many *final outputs*.

The TEEB service type, habitat services, is proposed by the EEA for CICES as part of a broader regulating and maintenance section. It is argued that habitat services capture aspects of natural capital that are important for the maintenance of physical, chemical, biological conditions in ecosystems (e.g. pest and disease control, pollination, gene pool protection) and are equivalent to other biophysical factors that regulate the ambient conditions such as climate regulation (Haines-Young and Potschin, 2013). The broad consultation process with the scientific community and practitioners for CICES, according to Haines-Young and Potschin (2013), confirmed the importance of making a clear distinction between final ecosystem services, ecosystem goods or products and ecosystem benefit. Therefore, the definitions listed in Table 4.3 have been recommended as the basis for CICES.

Term	Definition
Final ecosystem services	are the contributions that ecosystems make to human well-being. These services are final in that they are the outputs of ecosystems (whether natural, semi-natural or highly modified) that most directly affect the well-being of people. A fundamental characteristic is that they retain a connection to the underlying ecosystem functions, processes and structures that generate them.
Ecosystem goods and benefits	are things that people create or derive from final ecosystem services. These final outputs from ecosystems have been turned into products or experiences that are not functionally connected to the systems from which they were derived. Goods and benefits can be referred to collectively as *products*.
Human well-being	is that which arises from adequate access to the basic materials for a good life needed to sustain freedom of choice and action, health, good social relations and security. The state of well-being is dependent on the aggregated output of ecosystem goods and benefits, the provision of which can change the status of well-being.

Table 4.3: Definition of the terms *final ecosystem services*, *ecosystem goods and benefits*, and *human well-being* proposed for CICES (Haines-Young and Potschin, 2013)

The EEA's recommendation for CICES distinguishes ecosystem services, which are connected to underlying ecological structures and processes, from ecosystem products and benefits which are not connected to these structures and processes.

A key early decision in designing CICES was to exclude the so-called supporting services from the classification and focus only on the provisioning, regulating and cultural components. According to Haines-Young and Potschin (2013), the reason for the omission of supporting services was the need to identify and describe the *final outputs* from ecosystems that people use and value in order to link ecosystem and economic accounts in a corresponding manner. In doing so, the problem of *double counting* of ecosystem services (e.g. as supporting and regulating services) is supposedly avoided because the value of the ecological structures and processes that contribute to final outcomes is already wrapped up in their estimation (cf. Boyd and Banzhaf, 2007; Wallace, 2008; Fisher and Turner, 2008; Balmford et al., 2008). This implies that for the economic valuation of ecosystem services the contributions of these final services essentially matter for achieving benefits, while adding up the value of direct and indirect contributions to the same benefits made by the underpinning ecological functions can be avoided. This, however, does not subordinate supporting services (often also referred to as intermediate services). Supporting services may certainly be accounted for as underlying ecological structures, processes and functions which a number of different final services depend on, but in order to define an appropriate interface between ecosystems and society the exclusive use of *final services* has been proposed for CICES (Haines-Young and Potschin, 2013).

The notion of ecosystem services as final outputs has important implications for the valuation of ecosystems (see Section 4.3). Ecosystem functions based on ecosystem structures and processes provide ecosystem services but are only implicitly valued through the valuation of end products (Kroeger and Casey, 2007). This distinction is necessary for the assessment of ecosystem service alternation in provision and the rationale for the valuation of ecosystem services, for instance, through avoiding cost methods or substitution cost. It is also important to value the cost of service provision. Here, it is the improvement of ecosystem functions and processes that is important for final service provision.

The following example is useful to under the terminology introduced in the above. Ecological structures and processes of ecosystems determine the functions, i.e. the functioning itself, of ecosystems. Ecosystem functions, thus, describe the potential ecosystems have to provide ecosystem services. Nutrient cycling and soil permeability are an ecosystem function

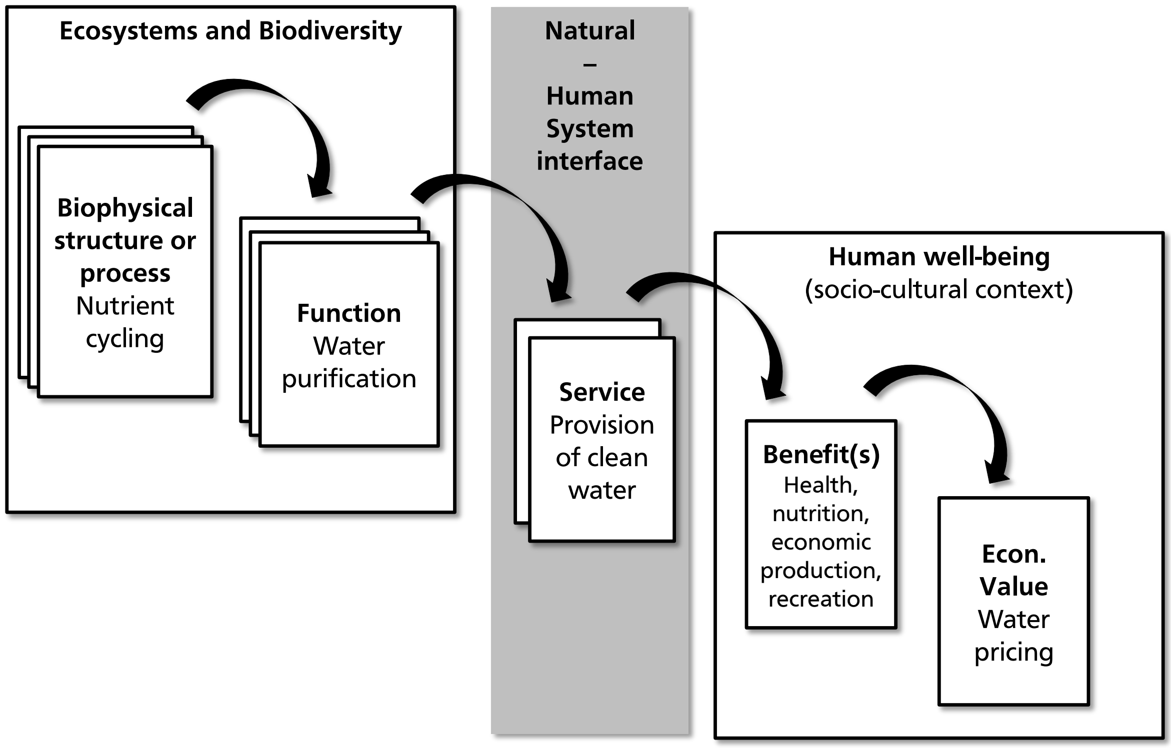

Figure 4.3: Illustration of the pathway from ecosystem structures and processes to human well-being; based on Haines-Young and Potschin (2010) and Maltby (2009) as presented in TEEB (2010a)

and an ecosystem process that is needed (possibly among other functions and processes) to purify water, i.e. an ecosystem function, to provide clean water as a provisioning ecosystem service. From this final ecosystem output, clean water, many different contributions to benefits can be derived, including the use of it for nutrition, pleasure or even for cultural identity. Finally, these benefits may also be valued economically, e.g. through estimations of Willingness To Pay (WTP) for a specific service or a bundle of them (see Section 4.3.1). This *pathway* from a natural system context of ecosystem structures, processes and functions over ecosystem services to a human system context of benefits and (economic) values is illustrated in Figure 4.3. Here, ecosystem services represent the interface between the natural and the human system.

It is important to note that just as the same ecosystem structures and process facilitate different ecosystem functions, so do ecosystem functions bear the potential to deliver different ecosystem services. These ecosystem services in turn may contribute to several benefits. The direct and indirect final outcomes (ecosystem services) are what people deem useful, thus, representing social preferences that may change over time and from place to place, regardless of ecosystem change. The complexity of ecosystem processes, structure and resulting functions as well as the role of biodiversity to provide redundancy for ecosystem resilience is still far from being clearly understood. Falkenmark and Rockström (2004), for instance, describe the role of biodiversity and resilience as follows: "While resilience is a buffer to disturbance, biological diversity plays an essential role in this buffer capacity - it acts as insurance by providing overlapping functions for restoring ecosystem capacity to generate essential ecological services".

When considering the impact of ecosystem change in order to improve the provision of desired services on other ecosystem service provision, it is important to realize that ecosystem services are often interdependent (Heal et al., 2001; MEA, 2005), and that the relationships between them may be highly non-linear (Farber et al., 2002; van Jaarsveld et al., 2005). Single ecosystem services, thus, represent different elements of an interrelated *bundle* (cf. Cumming et al., 2006). Therefore, optimization of a single service is often *traded-off* by reductions or losses of other services (Holling and Meffe, 1996). Figure 4.4 schematically illustrates the transition from ecosystem service provision of an ecosystem in a natural state (1) toward extensive (2) and intensive (3) agricultural use. In this context, for instance, it has been reported that regulating and cultural services are often higher in natural and semi-natural ecosystems than in heavier modified ecosystems (Pearce and Turner, 1990; de Groot, 1992; Costanza et al., 1997).

According to Rodríguez et al. (2006), trade-offs between the provision of different ecosystem services are very common and amply reported. Jackson et al. (2005), for instance, revealed trade-offs in the case of carbon sequestration and water provision services, while Kanowski and Catterall (2010) document trade-offs between carbon sequestration and biodiversity in Australian ecosystems. Hence, Muradian and Rival (2012) argue that a focus on single services has to be considered critically because of the complex and not yet well-understood structure of ecosystems.

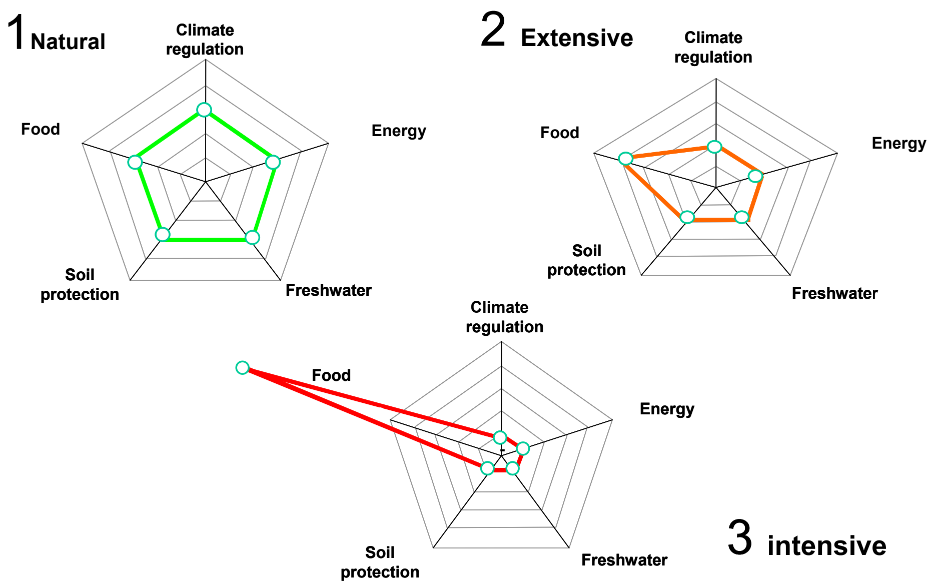

Figure 4.4: Transition of distribution of ecosystem service provision from an ecosystem in a natural state (1) toward an extensively (2) and intensively agriculturally used (3) one (Braat and ten Brink, 2008)

Without a doubt, the main driver of ecosystem change is human economic activity, most importantly, through agriculture and related land use changes (Vitousek et al., 1997). The management of natural resources generally aims at changing the composition and structure of ecosystems in forms more favorable for the provision of ecosystem goods for human well-being (Wallace, 2007). Human activities traditionally focus on an enhancement of on-site ecosystem service production (Foley et al., 2005), mainly in terms of increased agricultural output, thereby, intentionally or unintentionally, affecting the provision of other ecosystem services such as water regulation (MEA, 2003). However, the fact that landscapes simultaneously produce multiple ecosystem services that interrelate in complex dynamic ways has been largely overlooked (Chan et al., 2006; Rodríguez et al., 2006; Brauman et al., 2007). Unexpected or undesirable declines in other ecosystem services have resulted as unintended consequence of human domestication of ecosystems (Kremen and Ostfeld, 2005). On a global scale the MEA highlighted that the increase in a few services, such as food and timber, resulted in a decline in most other services such as flood control, genetic resources, or pollination (MEA, 2005). This has been one major reason for approaches of natural resource management focusing on the integration different environmental media and considering ecosystems as a whole, such as IWRM.

Although human-managed ecosystems, e.g. for agriculture, are driven by human management (e.g. soil tillage, irrigation, nutrient additions), they are still influenced by the same ecological processes that shape and drive natural ecosystems, particularly by those processes that support biomass production and others such as nitrogen uptake from the atmosphere and pollination of crops (Falkenmark and Finlayson, 2007). Ecosystems modified as agricultural systems are often referred to as agroecosystems (Falkenmark and Finlayson, 2007). Today the difference between a natural ecosystem and an agroecosystem is related to the extent of human intervention or management and considered to be largely conceptual.

Rather recently these external effects, i.e. externalities gained more importance, for instance, in the context of global climate change or biodiversity loss. The ecosystem service concept has served as an eye-opener (cf. Costanza et al., 1997; Gordon and Folke, 2000) for traditionally undervalued ecosystem services and connects ecosystems with on-site - and more importantly - off-site beneficiaries. As humans are altering the potential of the ecosystem to provide services though different land uses, the ecosystem service concept connects land users with beneficiaries. Therefore, land users may be regarded as providers of ecosystem services and disservices. The interrelationships between different ecosystem services, thus, are an important issue. A management focus on individual ecosystem services, as standalone services, may lead to trade-offs among services resulting in unwanted declines in some ecosystem services (MEA, 2005) or even regime shifts with unexpected consequences (Gordon et al., 2008). However, these trade-offs may possibly be altered by focusing on managing the ecosystem processes that link services (Pretty et al., 2006). However, despite the growing awareness of the socio-economic relevance of ecosystem services, actual mainstreaming and implementation of ecosystem services in practical planning and decision-making can be considered as still in its infancy (Naidoo et al., 2006; Daily et al., 2009).

The ecosystem service concept has been addressed by different economic schools of thought of environmental and ecological economists including some rejecting it entirely. Therefore, Farley and Costanza (2010) present a particular definition of ecosystem services and ecosystem goods as an attempt to reconcile the contrasting conceptualization of environmental economic, for instance as introduced by Engel et al. (2008) and ecological economic approaches, discussed by Muradian et al. (2010), as well as those rejecting the concept of ecosystem services because of the threat of commodification of ES, as expressed for instance by McCauley (2006); Robertson (2006); Kosoy and Corbera (2010). While the environmental economics approach prioritizes economic efficiency by integrating ecosystem services into the classical market model, the ecological economics approach focuses more broadly on the multiple goals of ecological sustainability, just distribution and economic efficiency favoring a variety of valuation mechanisms to achieve these goals, both market and non-market based. Accordingly, in the ecological economics approach appropriate institutions and mechanisms are determined by and adapted to the relevant characteristics of the ecosystems and services in question. In presenting a distinct definition for ecosystem services and goods, Farley and Costanza (2010) aim to unify all three perspectives in order to identify appropriate institutions for their governance based on the physical characteristics of the services in question. Moreover, the definition presented is based on the reasoning that ecosystem services are essential, non-substitutable and poorly understood, while there are real costs to their provision and protection. Since someone has to pay those costs, this reasoning justifies ecosystem service valuation without requiring commodification of ecosystem services. Thus, Farley and Costanza (2010) pick up a unifying feature of most ecosystem service definitions that define services as processes or functions of value to humans and elaborate a definition more suitable for the analysis of policy instruments such as Payments for Ecosystem Services (introduced in Chapter 5). However, Farley and Costanza (2010) argue that payment mechanisms based on pure market logic will only rarely be appropriate.

In their definition, Farley and Costanza (2010) propose to differentiate between ecosystem goods as *stock-flow resources* and ecosystem services as *fund-services* (as introduced by Georgescu-Roegen in 1971) provided by nature, thus connecting the economic subsystem to the ecosystem. Ecosystem goods, i.e. *stock-flow resources*, represent the physical flow or throughput of raw materials and stored energy from nature transformed into economic products and later returned to nature as disordered waste. These goods, equally considered as ecosystem structures (Daly and Farley, 2010), may be used at the rate society chooses transforming them physically in the act of production, e.g. forests into houses. Hence, according to Farley and Costanza (2010) they can be quantitatively used up, but also be stockpiled when inflows exceed outflows.

In contrast, ecosystem services are described as *fund-services* having fundamentally different physical characteristics. *Fund-services* are functionally connected to *stock-flow resources* since they are generated by a particular configuration of *stock-flow resources* (Farley and Costanza, 2010). As opposed to ecosystem goods, the use rate of ecosystem services cannot be accelerated because ecosystem services are available at a given rate over time. The generation of ecosystem services does not transform the provisioning ecosystem; instead of being used up quantitatively quantitatively the ecosystem is changed qualitatively. A further difference to ecosystem goods is that ecosystem services cannot be stockpiled.

Farley and Costanza (2010) present five conceptual advantages of their definition:

- Defining ecosystem services as fund-services reflects the valuation of land uses associated with generating the service, i.e. the link between land use and service provision, more explicitly.
- The definition encompasses existing definitions, e.g. the one by Fisher and Turner (2008) emphasizing the fund side and the MEA's definition focusing on the service side, stressing that these two actually work together. The definition describes provisioning service as the capacity of ecosystem structure to reproduce itself, rather than the food, fiber, fuel and water provided representing ecosystem goods.
- Ecosystem services are described as the physical characteristics of fund services rather than their explicit value to humans. This takes into account the limited knowledge and understanding of ecosystem functioning which might be of value for humans. Hence, defining ecosystem services by their physical characteristics stresses that they do not blend readily with market institutions, and other economic institutions may be required to protect and provide them.
- Defining ecosystem services as generated by a particular configuration of stock-flow resources emphasizes their role as an emergent property of complex systems.
- In distinguishing between stock-flow and fund-services, the dual nature of natural resources is highlighted. This may help to analyze why market economies systematically favor the conversion of ecosystem structure into stock-flow inputs and into economic production over its conservation in ecosystem funds in order to provide ecosystem services (Farley, 2010). Moreover, as stock-flow and fund-service resources are typically complements it stresses the limits of their substitution.

With their definition of ecosystem goods and services, Farley and Costanza (2010) stress the need to conserve the dual functions of ecosystems, as stock-flows that could provide raw materials for economic production and as fund-fluxes that provide critical ecosystem services. By using up ecosystem goods in form of stock-flows, the structure of ecosystems may be converted to economic output, and ecosystem functions are sufficiently compromised. This is a trade-off that has to be considered.

When considering the ecosystem service concept in the context of IWRM the focus has to be laid on so-called hydrological ecosystem services. This type of services will be introduced in the following section.

4.2 Hydrological Ecosystem Functions and Services

The provision of hydrological or water-related ecosystem services is one element of a bundle of other essential services - such as air quality, carbon dioxide sequestration, and soil generation - provided by ecosystems which are interrelated in dynamic and complex ways. Some authors refer to these services by using the term *watershed services* (Smith et al., 2006), thus, highlighting the use of the concept in the context of water management. A reasonable distinction of different water-related services is provided by Brauman (2009) in dealing with aquatic services representing direct services from freshwater ecosystems themselves, e.g. in-stream nutrient removal, and hydrological services describing the terrestrial effects on hydrological flows separately. Hence, the quality, quantity and timing of water draining into and flowing along rivers is modified by topography, geology, soil type, vegetation cover, land use and other human activities within a river basin. Primarily, water is partly returned to the atmosphere via evaporation from lakes, wetlands, the soil surface and wet vegetation, and through transpiration by plants and trees but another part of water moves down slopes and stream channels, as well as underground, carrying sediment, nutrients and other chemicals or contaminants. Thus, the quality, quantity and timing of water available to downstream users in a river basin depends on the particular types and distribution of vegetation, the underlying geology, the soil types present and the way that land is used and managed. Therefore, watershed services, i.e. hydrological ecosystem services - as they are referred to - in this dissertation, are final outputs of ecosystem functions or processes that provide different kinds of direct and indirect streams of benefits to humans in the following main categories (cf. Tognetti et al., 2006):

- Provision of freshwater for:
  - Consumptive uses (drinking, domestic, agricultural, and industrial)
  - Non-consumptive uses (hydropower generation, cooling water, and navigation)
- Flow regulation and filtration, key aspects of which are the control of mean surface runoff, peak or flood flows, base or dry season flow, and erosion and sediment load, as well as recharge of groundwater and soil moisture (FAO, 2002). Benefits of these may include the following:
  - Water storage in soils, wetlands, and floodplains, which can buffer flood flows and drought
  - Control of erosion and sedimentation which, in excess, can have adverse effects on aquatic life, irrigation canals, dams, and navigation. Below normal flows of sediment downstream from dams can have adverse effects on coastal areas where it provides protection from erosion and nourishes the development of mangroves, both of which can reduce storm damage
  - Maintenance of river channels, wetlands, riparian habitats, fisheries, and other wildlife habitat that may be important for hunting, migratory birds, rice cultivation, and fertilization of floodplains
  - Maintenance of mangroves, estuaries, and coastal zone processes, which often rely on seasonal pulses of freshwater inputs, and are critical habitats for fisheries as well as for other marine life
  - Control of the level of groundwater tables that may have adverse effects on agriculture by bringing salinity to the surface.
  - Maintenance of water quality, which may be impacted by inputs of nutrients and organic matter, pathogens, pesticides, and other persistent organic pollutants, salinity, heavy metals, and changes in the thermal regime

Apart from these provisioning and regulating services there are also important hydrological supporting services, including the maintenance of natural flow and disturbance regimes as drivers of ecosystem processes which support aquatic ecosystem resilience, often referred to as *environmental flows*, as well as services that support cultural values, e.g. aesthetic qualities that support tourism and recreational uses (Tognetti et al., 2006). The interdependence of all of these services often implies trade-offs between the provision of freshwater for direct uses and the regulatory and supporting services that insure its continued provision.

Brauman et al. (2007) define hydrological services more narrowly as those ecosystem services which "encompass the benefits to people produced by terrestrial ecosystem effects on freshwater". This definition focuses explicitly on the off-site final services. Brauman et al. (2007) organize these services into five broad categories:

- Improvement of extractive water supply
- Improvement of in-stream water supply
- Water damage mitigation
- Provision of water-related cultural services, and
- Water-associated supporting services.

Brauman et al. (2007) provide three aspects related to hydrological ecosystem services to describe the pathway from service provision to the final outputs that beneficiaries value (see Figure 4.5): ecohydrologic processes, hydrologic attributes and hydrological services themselves. In a management and policy context it is useful to consider these three aspects since they convey specific information to facilitate institutional fit and interplay. The first aspect, ecohydrological processes (and structures) describes the provisioning function of ecosystems and therefore corresponds to the providers of

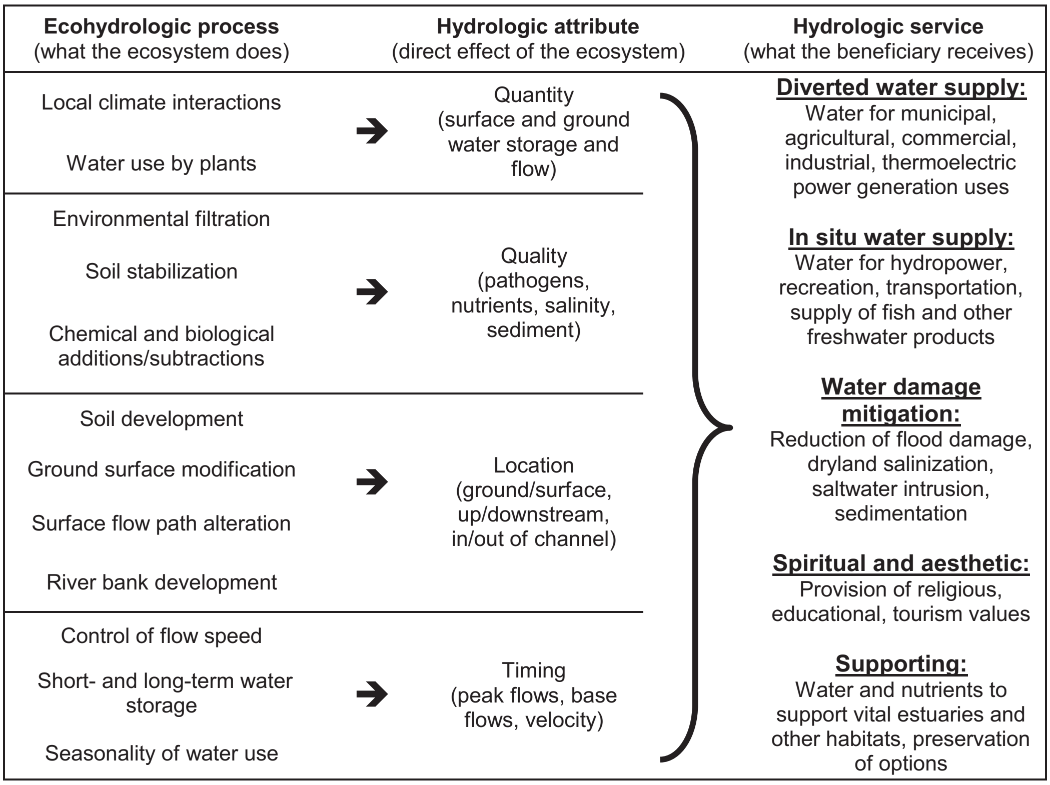

Figure 4.5: Relationship between hydrological ecosystem processes and services (Brauman et al., 2007)

on-site and off-site ecosystem services. Different ecosystems may have similar functions (e.g. water retention of woodlands and wetlands) to similar but also to different degrees.

The second aspect, hydrological attributes, describes qualitative and quantitative as well as temporal and spatial characteristics affected by changes in ecohydrological functioning. Each and every hydrological ecosystem service is determined by certain hydrological attributes. The requirements on hydrological attributes can vary for different hydrological ecosystem services, thus, trade-offs are common here as well. Moreover, each hydrological attribute is directly impacted, improved or degraded, by ecosystem processes and structures as water moves through a landscape. While flowing through an ecosystem, different ecosystem processes may have competing effects on the same attribute or simultaneously have positive and negative effects on different attributes of a particular service. Thus, infiltration may be increased in a forest ecosystem, for instance, while total liquid water volume leaving the ecosystem may be decreased through transpiration (Brauman et al., 2007). The hydrological attributes of a provisioning hydrological ecosystem service as a potential benefit for off-stream use, e.g. municipal water supply, are defined by a specified volume of water of an expected quality in a certain form and at a certain time. Other beneficiaries, those of flood damage mitigation, are likely to have less formal but similarly important requirements. Figure 4.6 illustrates how different processes and structures of ecosystems relate to hydrological attributes.

The latter aspect of the conceptualization of hydrological ecosystem services by Brauman et al. (2007) refers to the final benefits and beneficiaries of services (although the authors call them hydrologic services, according to the definitions presented in Section 4.1 these are actually benefits). These benefits and beneficiaries represent (potential) direct and indirect water users. Together with the providers of ecosystem services they represent the stakeholders across all sectors that IWRM intends to integrate. However, they are not considered by sector, instead the ecosystem (in the IWRM context it is commonly the water resource itself represented as a “comb” that connects the different sectors and stakeholders) as the basis for all (economic) activity serves as the principal integrator of the natural and human system.

Since any ecosystem within a river basin affects the attributes of the water that passes through it, all ecosystems provide hydrological services, although to differing degrees (Allan, 2004). All elements of an ecosystem, from microbes to megafauna, affect hydrological service provision, but vegetation, i.e. land use and land cover, is often the driving force in ecosystem effects on water, especially when considering short and medium term disturbances. Contrary to other ecosystem services, e.g. carbon sequestration, hydrological ecosystem services are generally local and regional services are defined by the biogeophysical properties of a river basin and its sub-basins. Hence, downstream users experience the effects of ecosystem change throughout their river basin. At different spatial and temporal scales different (internal and external)

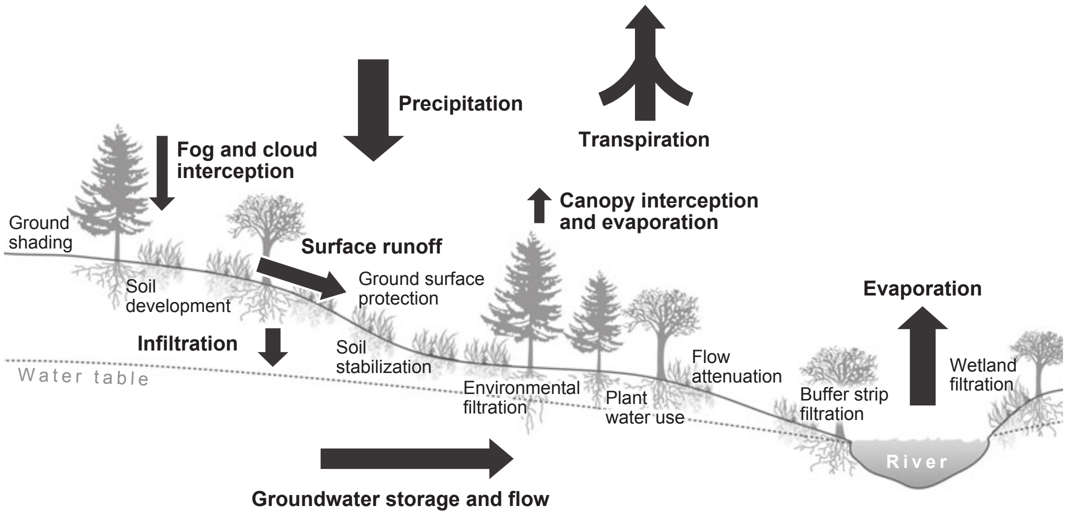

Figure 4.6: Water cycle-ecosystem interactions. Arrows indicate fluxes of water (Brauman et al., 2007).

drivers of change gain and loose importance. For instance, the impact of land cover change (internal driver) may be diffused in larger river basins while a large-scale meteorological event can cover larger parts (Birkinshaw and Bathurst, 2006). According to Brauman et al. (2007), for instance, the ecosystem effects on sediment yield and flooding are only measurable in small river basins and for small rainfall events. Depending on the dimension and location of different ecosystems within the basin and on the frequency, duration, and intensity of climatic events hydrological ecosystem service effects may either decrease or increase with basin size, extrapolations of local and short-term effects of hydrological services to larger scales may therefore be flawed (Brauman et al., 2007).

The Food and Agriculture Organization of the United Nations (FAO) investigated the impacts of land use on hydrological and landscape processes on the basis of a large number of hydrological studies and came to similar results (FAO, 2002). According to the FAO, the results, summarized in Table 4.4, provide some “general rules“ indicating that impacts can only be verified within small basins, and most of the hydrological studies in fact pertain to small-scale river basins. Since at larger scales, natural processes (e.g. meteorological phenomena) are dominant, it is difficult to detect any change as a result of conservation practices, particularly on a short time scale. In contrast, impacts of land use on water quality can be observed at much larger scales. In some cases, water quality impacts have been well documented and quantified even in larger basins (FAO, 2002). In case of various sources of pollutants, however, a direct linkage of causes and effects may remain elusive.

For the identification of hydrological service provision and related benefits, information on the scale at which land use practices have a verifiable impact on hydrological attributes (water resources availability in volume and timing as well as quality) is crucial. This information defines the spatial fit and institutional interplay of provider and beneficiary constellations and determines the feasibility of policy intervention toward a more equitable sharing of costs and benefits. The latter is only the case when impacts of land use can actually be linked with downstream effects to a reasonable degree of uncertainty.

Recognizing that the dynamics of hydrological ecosystem service interactions are crucially influenced by the scale at which the phenomena are studied as well as by the physiological characteristics of vegetation, the pedology of the soil, and the type of climate, a new field of studies emerged with ecohydrology in the late 1990s. Rodriguez-Iturbe (2000) defines ecohydrology as “the science which seeks to describe the hydrologic mechanisms that underlie ecologic patterns and processes”. Moreover, ecohydrology quantifies and explains the relationships between hydrological processes and biotic dynamics at a catchment scale (Zalewski and Wagner-Lotkowska, 2004). Hence, ecohydrology incorporates the ecosystem service approach but carries it further in the context of ecological engineering. Thus, it considers how ecological processes and structures can be used and enhance through “hydro-technical infrastructure“ (cf. Zalewski et al., 1997) in order to improve hydrological ecosystem service provision. A central principle of ecohydology is to use ecosystem properties as management tools at the river basin scale Zalewski (2000). The climate-soil-vegetation dynamics (and their alternation by management practices) are considered as the core of hydrology with the soil moisture balance, in absence of pronounced topographical effects, as the principle determinant for infiltration, evaporation and leakage quantities (Sandström, 1998; Rodriguez-Iturbe, 2000; Brauman et al., 2007).

The ecohydrological perspective on land use changes in a river basin focuses on their effects on water flows. The link between land use and the provision of hydrological ecosystem services is found in the partitioning points of the water cycle.

Impact Type[1]	Approximate basin size [km²] 0,1	1	10	10^2	10^3	10^4	10^5	Source
Average flow	X	X	X	X	–	–	–	(Bosch and Hewlett, 1982; Chapman and Falkenmark, 1989; Bruijnzeel, 1990)
Peak flow	X	X	X	X	–	–	–	(Bruijnzeel, 1990; Brooks et al., 1991; La Marche and Lettenmaier, 2001)
Base, dry season flow	X	X	X	X	–	–	–	(Brooks et al., 1991; Calder, 1998; Chomitz et al., 1999)
Groundwater recharge	X	X	X	X	–	–	–	(Tejwani, 1993; Calder, 1998; Chomitz et al., 1999)
Sediment load	X	X	X	X	–	–	–	(Chapman and Falkenmark, 1989; Bruijnzeel, 1990; Brooks et al., 1991)
Nutrients	X	X	X	X	X	–	–	(Chapman and Falkenmark, 1989; Brooks et al., 1991)
Organic matter	X	X	X	X	–	–	–	(BGS, 1996; Ongley, 1996)
Pathogens	X	X	X	–	–	–	–	(Dougherty and Hall, 1995)
Salinity	X	X	X	X	X	X	X	(BGS, 1996; FAO, 1997)
Pesticides	X	X	X	X	X	X	X	(Ongley, 1996)
Heavy metals	X	X	X	X	X	X	X	(Ongley, 1996)
Thermal regime	X	X	–	–	–	–	–	(Westcot, 1997)

Table 4.4: Measurability of land use effects by basin size; based on FAO (2002)

[1] Legend: X = Measurable impact; – = No measurable impact

Any kind of land use and its interaction between soil, water, plant and atmosphere will generate a specific partitioning of rainfall. Falkenmark and Rockström (2004) conceptualize the climate-soil-vegetation dynamics with their green and blue water approach, while focusing on two principal partitioning points of precipitation. The main biophysical and human determinants for this partitioning are summarized in Table 4.5.

Flow component	Biophysical determinant	Human determinant
1st Partitioning point		
Surface runoff	Vegetation / Biomes	Land use
	Soil surface conditions	Tillage practices
	Rainfall intensity	–
	Soil wetness	Soil management
Soil moisture	Water holding capacity in soil	Soil management
2nd Partitioning point		
Evaporation	Atmospheric demand (potential evaporation)	Canopy cover
	Micro-meteorology	Mulching
	Wetness of soil	Timing of planting
Transpiration	Photosynthetic pathway	Crop management
	Plant available soil moisture	Forest management
	Atmospheric demand	–
Groundwater recharge	Soil hydraulic conditions	Compaction
	Geological conditions	

Table 4.5: Biophysical and human factors that determine the partitioning of water flows in the hydrological cycle (Falkenmark and Rockström, 2004)

Additionally to the biophysical determinants of rainfall partitioning, land management has a primary influence as the most important human determinant. Falkenmark and Rockström (2004) point out that rainfall partitioning significantly determines water flow paths at various scales. Land use and natural conditions, therefore, influence flow partitioning on all scales, from the individual plant to the river basin with considerable impacts on on-site and off-site water availability. This is a strong argument to shift the conventional water resource management focus on blue water alone, which is directly associated with stable blue water flow in rivers and aquifers downstream in river basins, toward more consideration of green water flows that influence the rainfall partitioning upstream which in turn determines blue water flows downstream. Hence, "rainfall, not stable blue water flow, is the fundamental water resource" in the concept of ecohydrology (Falkenmark and Rockström, 2004).

The ecohydrology concept is particularly useful to understand the provisioning side of hydrological ecosystem services by providing a land use - water flow path consideration. Moreover, it illustrates the natural determinants of the hydrological cycle and the possible human influence on those. Hence, land users can be considered as contributors to the provision of ecosystem services, i.e. disservices. Thus, the concept facilitates comparisons of land use impacts and natural factors on service provision, at least at the primary partitioning points of rainfall. While crossing the river basin either on the surface or the subsurface, the hydrological attributes are further determined by additional processes. For superficial, i.e. overland, water flows the presence of buffer strips or re-infiltration areas are important for flow retention, erosion control and sediment reduction before water reaches a stream or river reach. The interaction of subsurface flows and rivers can be of effluent or influent character, determining the flow in different ways. As pointed out before, in-stream processes subsequently alter hydrological attributes. Each of these processes within river basins may vary significantly from one climatic zone to another.

In the context of human determinants of service provision, Falkenmark and Rockström (2004) highlight overgrazing, deforestation and land mismanagement in agriculture as three major causes of human-induced land degradation. They all cause a shift toward increased erosive surface runoff, reduced productive green water flow (biomass production) and lowered ground water recharge as a result of their effect on rainfall partitioning. Deforestation, for instance, may cause more storm water surface flow and less groundwater recharge, and may turn a relatively water-rich region into a region experiencing seasonal water scarcity (Agarwal, 2000; Falkenmark and Rockström, 2004). Although many aspects of hydrological response are often determined by extreme but infrequent meteorological events (cf. Brauman et al., 2007), the ability of ecosystems to mediate these is unclear. This mediation of extreme events is, thus, of considerable importance and presumably not linearly related to the delivery of water services in average years. Most of the knowledge about ecosystem effects on hydrological ecosystem service provision, e.g. water supply and water hazard mitigation, stems from research activities conducted in temperate ecosystems (Brauman et al., 2007). Hydrological responses for a different climate, geography, or ecosystem type have been studied to a lesser degree.

Meybeck (2003) realized a global analysis of human impacts on river basins and demonstrated that the modification of aquatic systems by human pressures (e.g., flood regulation, fragmentation, sedimentation imbalance, salinization, contamination, eutrophication, etc.) has increased to a level that can no longer be considered as being controlled by natural processes (climate, relief, vegetation, limnology) only. Based on these conclusions, Zalewski and Wagner-Lotkowska (2004) regard technical solutions alone as insufficient to reduce or eliminate the discharge of pollutant, instead tools are necessary to manage the degradation of ecological processes in landscapes based on an understanding of the temporal and spatial patterns of catchment scale water dynamics.

In the context of the role that different ecosystems play for the provision of hydrological ecosystem services there has been a big discussion, especially on the role of forest ecosystems. This issue has been addressed repeatedly by Salati and Vose (1984); Calder (1998); Sandström (1998); Calder (2002); Aylward (2004); Bonell and Bruijnzeel (2004b); Kaimowitz (2004); Bonell and Bruijnzeel (2004a); Andréassian (2004) and several other authors.

The principal experimental method to investigate the impact of land use changes on hydrological flows is paired catchment studies (cf., Bosch and Hewlett, 1982; Brown et al., 2005). These studies generally consider four broad categories: afforestation experiments, deforestation experiments, regrowth experiments and forest conversion experiments. The principal advantage of paired catchment studies is that macro-scale and meso-scale climate variability is removed through the comparison of two catchments subject to similar climatic conditions under different land uses Brown et al. (2005).

The subsequent land use and the respective degradation of hydrological functions is important for the resulting altered hydrological response. Overall annual water yield often has limited explanatory power since the total water yield may increase because of high amounts of surface runoff during the wet season, while dry season flow may decrease because of lower infiltration during the wet season (Bruijnzeel, 1988). Roa-García et al. (2011) provide evidence for this *infiltration trade-off hypothesis* (cf. Bruijnzeel, 1988, 2004) for tropical environments (the Andes), which assumes that after forest removal, soil infiltration rates are smaller and the water losses through quick flow are larger than the gains by reduced evapotranspiration. This hypothesis has been reaffirmed recently by Bonell et al. (2010), Krishnaswamy et al. (2012) and Krishnaswamy et al. (2013) for several Indian river basins. Brown et al. (2005) conclude similarly with regard to their comparison of paired catchment studies that “the difference in hydrological responses observed in tropical summer dominant rainfall catchments, undergoing similar changes in vegetation highlights the difficulties associated with making generalisations about the seasonal impacts on water yield. At the seasonal time scale other catchment characteristics such as soil depth and type play a much larger role in the response than on the mean annual basis“.

The role of forested areas in the tropics for the maintenance of dry season flows was studied by Roa-García et al. (2011). The authors revealed, in a paired catchment experiment that the basin with the highest forest cover (68%) showed the smallest reduction in flow during the dry season and the highest low flows were maintained there, even when compared to grassland dominated basins. Moreover, it could be proven that natural forests have a larger capacity to store and release soil moisture than grassland (Roa-García et al., 2011; Krishnaswamy et al., 2013).

However, in order to record changes in total annual water yield around 20% of the catchment area had to be altered in most paired catchment studies, but there are cases where only 15% (Rocky Mountains) or up to 50% (Great Plains) had to be changed to cause any response (Brown et al., 2005).

The limitations of paired catchment studies have been addressed by Bruijnzeel (2006) who refers to them as a time-consuming method (typically more than 5 years) and thus an expensive affair. Moreover, he regards the method as being essentially a "black-box" requiring additional process research to reveal the relative importance of different causative factors to explain the changes observed in streamflow. Both aspects have led, according to Bruijnzeel (2006), "to a general decline in the number of such studies in the last few decades and a gradually increasing emphasis on computer simulations". Nevertheless, some lessons learned from the limitations of paired catchment experiments can still be derived for general consideration with regard to hydrological responses due to land use changes:

- The results of permanent vegetation change experiments indicate that, depending on the changes in soil storage and the transpiration-vegetation age characteristics of the new vegetation type, it takes longer than 5 years for a new hydrologic equilibrium to be established.
- Changes in vegetation type will affect not only mean annual flow, but also the variability of annual flow.
- Flow duration curves provide a useful means of displaying the complete range of daily flows and allow the impacts on low and high flows to be assessed at different temporal scales (annual or seasonal). Seasonal flow duration curves can be used to assess the seasonal impacts on daily flow (Brown et al., 2005).

In general, it seems inappropriate to make generalizations on the effects of land use changes on hydrological responses only on the basis of land uses in the sense of land cover, e.g. forest changed into pasture. Most important is how the different rainfall partitioning points are quantitatively and qualitatively changed through the change in vegetation and soil properties which together influence the critical processes of evapotranspiration, infiltration and soil moisture retention. These processes in turn determine whether Hortonian, infiltration-excess because of soil saturation or sealing, overland flow (cf. Horton, 1945) occurs which causes faster runoff responses and increased erosion. Figure 4.7 provides a schematic representation of the occurrence of various streamflow generating processes in relation to their major controls.

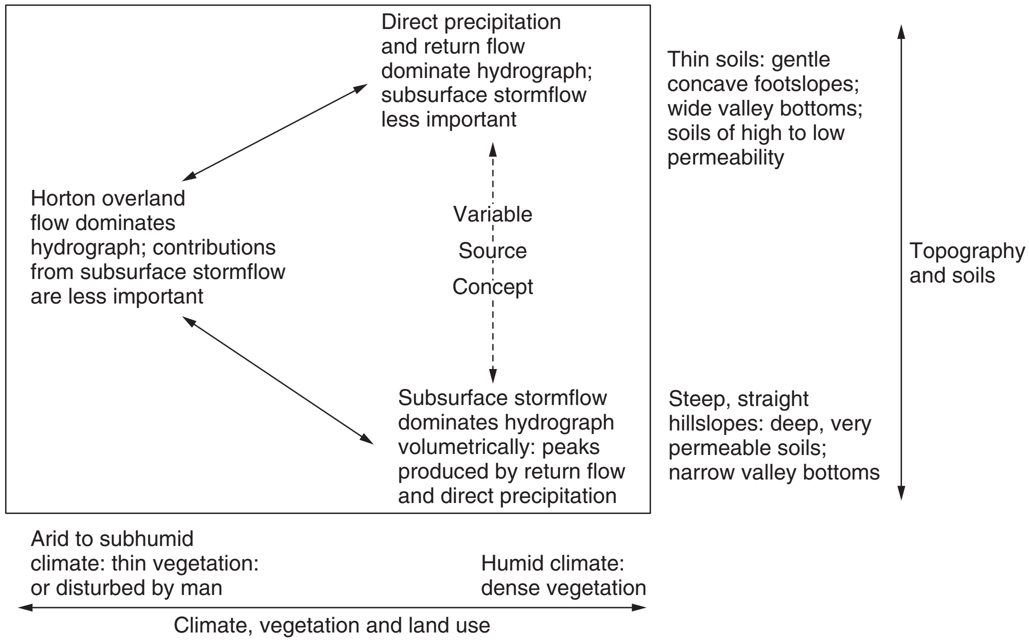

Figure 4.7: Schematic representation of the occurrence of various streamflow generating processes in relation to their major controls; from Bruijnzeel (2006) as reproduced from Dunne (1978).
Note: "direct precipitation and return flow" are equivalent to saturation overland flow

An important cause for infiltration excess occurrence is the reduction of hydraulic conductivity, a principle soil property (cf. Bouwer, 1966), as a result of degradation, e.g through top soil erosion or soil compaction (Krishnaswamy et al., 2013). Thus, it is more appropriate to consider the degradation degree compared to a former state in order to project possible hydrological responses. Furthermore, hydrological responses, especially those with longer time lags like seasonal and base flows, can also be significantly controlled by the geological properties of underlying bedrock. The annual precipitation distribution and the corresponding soil moisture conditions may also influence the variability of annual runoff (Peel et al., 2001).

Hydrological ecosystem service		Principal beneficiaries
Provision of freshwater	*in-stream*	Hydro power generation, transportation, water recreation, freshwater fish and other food production
	off-stream (extractive)	Municipal water supply, irrigation agriculture, commercial and industrial water user, thermoelectric power use
Regulating services		Water supply, flood-prone inhabitants and industries, agriculture, hydro power generation, transportation, water recreation, freshwater fish and other food production
Cultural services		Spiritual uses, aesthetic appreciation, tourism

Table 4.6: Hydrological ecosystem services and related beneficiaries; based on Brauman et al. (2007)

These insights call for being cautious with oversimplifying hydrological responses based on land use changes alone, instead a focus on alternation of hydrological process is more reasonable. This provides opportunities for considering different management practices, in addition to different land uses, in order to improve hydrological ecosystems service provision.

However, the spatial and temporal distribution of subsurface hydrological processes still represents a significant knowledge gap (Bogaart et al., 2011). These processes are characterized by large variability and heterogeneity (preferential flows, transient water tables etc.), but good observation methods are missing or in their infancy.

Besides stressing the supply side of hydrological ecosystem services and the role of changes to ecosystems by humans, the ecosystem service concept establishes a link to potential beneficiaries. Table 4.6 summarizes hydrological ecosystem service and related beneficiaries, i.e. water users.

Just as the supply side of hydrological ecosystem services provision comprises different sectors such as agriculture, forestry, nature conservation, so does the beneficiary side as well. The broad variety of different beneficiaries encompasses all water using sectors.

4.3 Economic valuation of Ecosystem Services

All Ecosystem Service definitions have a common anthropocentric perspective since their individual assessment is only made possible by a socio-economical valuation of ecological structures and processes (de Groot et al., 2002; Fisher et al., 2009; Haines-Young and Potschin, 2010). Therefore, ecosystem services can only be assessed in a defined societal context and may be valued (monetarily or non-monetarily) very differently from time to time and from place to place (Boyd and Banzhaf, 2007; Wallace, 2007). Thus, social preferences of ecosystem processes and functions are as important as their ecological interaction (Haines-Young and Potschin, 2010). Although grounded in a shared scientific methodology, the process of valuation and assigned economic values themselves are culturally and socially constructed, just as are the concepts of ecosystems, ecosystems services and biodiversity. Therefore, economic values reflect socially and culturally constructed mindsets, worldviews and realities of particular sectors of a society, hence, economic values do not represent universal truths or static objective facts (Wilk and Cliggett, 2006). Moreover, they are not exogenous to the valuing society, but instead reflect the everyday social interactions as well as political and power relationships within a system of interdependence of local, regional and global extent (Hornborg et al., 2007).

With the rise of the new environmentalism in the 1960s a shift in environmental concerns from the protection of empty spaces and particular species toward concerns with the human environment came about. The most prevailing outcome of this process has been the global establishment of protected areas (Zimmerer, 2006). Even today, this environmental policy results in more emphasis being placed on protecting and isolating ecosystems from economic development or commodity markets, than on redefining and regulating the relationship between the economy and the natural environment (TEEB, 2010a). A more direct effort to integrate the environment into the economy was identified in the Brundtland Report (WCED, 1987) by opening space for new conceptions based on the principle of inter-generational responsibility. The Convention on Biological Diversity (CBD), a key outcome of the Earth Summit of 1992, reflected the shift from species-based conservation to the conservation of ecosystems and biomes, while also highlighting the role of local populations as stewards for nature and a source of knowledge relevant to conservation and sustainable development (UN, 1992). Furthermore, the convention has put significant emphasis on the value, including economic value, of biodiversity and ecosystem services as well as local knowledge, e.g. through bioprospecting.

Further important contributions to integrate the environment into policy and economic thinking are represented in the Millennium Ecosystem Assessment (MEA) of 2005. While proposing a utilitarian and anthropocentric approach, with the

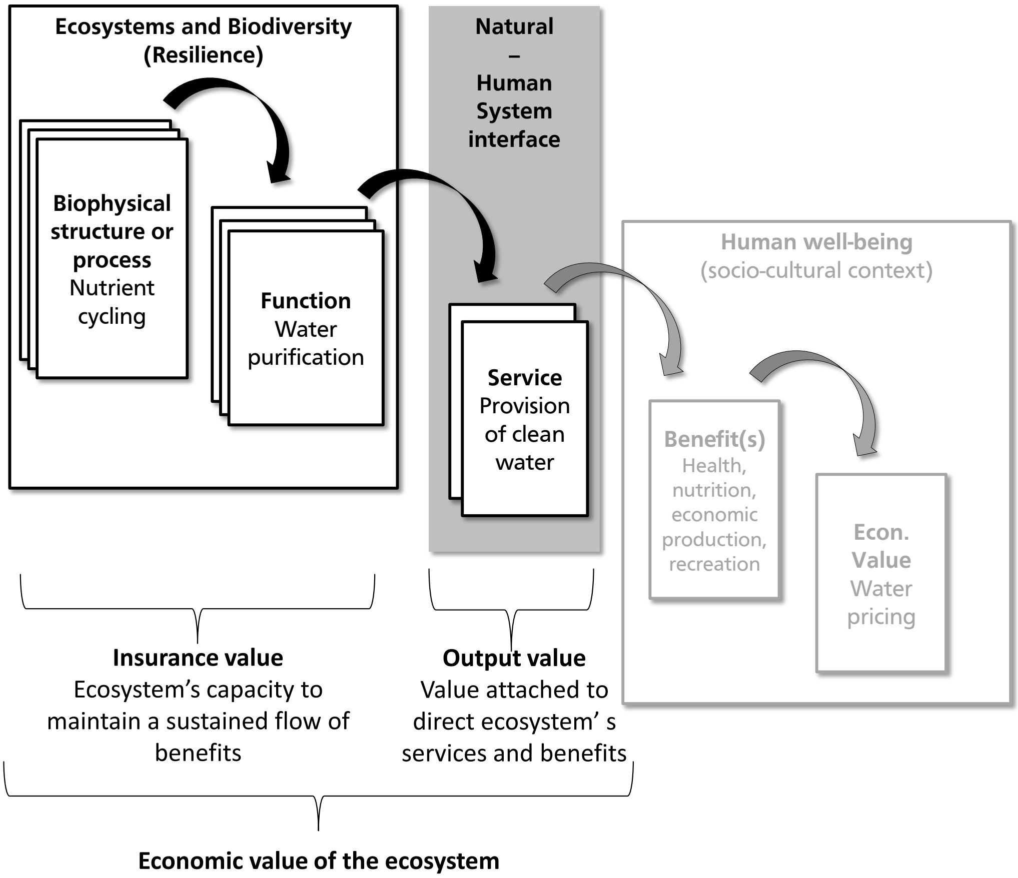

Figure 4.8: Ecosystem value composed of insurance and output value; based on TEEB (2010a)

ecosystem service concept as its core concept, it also emphasizes the dependency of humans not only on resources but on ecosystem functioning itself. Through the MEA's framework of human well-being depending on ecosystem services, a major contribution was made to the visibility of a broad array of ecological and biophysical functions usually taken for granted by society (MEA, 2005). This has also led to a much broader understanding of the global scale of human impact on the environment and the resulting economic and social consequences.

Meanwhile, the challenges of valuing biodiversity and ecosystem services have been widely recognized and the multi-dimensional, strongly context-dependent nature of valuation has been addressed by several authors (Turner et al., 2003; Shmelev, 2008; EPA, 2009). With the TEEB initiative, the economic valuation of biodiversity and ecosystem services was addressed even more explicitly.

However, despite these advances important limitations in economic valuation remain with respect to the inclusion of cultural and social values that need to be recognized (Shmelev, 2008). This has some implications for valuation, especially with regard to ethical concerns about what should and what should not be valued (cf., Turner et al., 2003; TEEB, 2010a).

Moreover, economic valuation of the environment has an important consequence in the way it contributes to change the notion of property and ownership described by Polyani (1944) as *commodity fiction*; see also the previously mentioned criticism by McCauley (2006); Robertson (2006); Kosoy and Corbera (2010). Thus, it is argued that through economic valuation, Ecosystem Services may be perceived as environmental commodities which "can be owned and traded in the market system for dollar" (Vatn and Bromley, 1994) while neglecting their intrinsic values. However, the lack of previously defined monetary values for non-marketed goods as most ecosystem services can lead to valuation processes that trigger negotiations between and within exogenous and endogenous systems of value (Cummings et al., 1986; Sagoff, 1988; Hanemann, 1994). Hence, besides the risk of a commodity fiction, economic valuation induces previously ignored monetary appreciations and new utilitarian frames of appreciation (TEEB, 2010a). How individuals assign economic values is yet far from being understood completely. Findings in behavioral economics suggest that utility and emotions are interdependent: "utility arises from emotions and emotions arise from changes, and people's judgment and choices have more intuitive than rational or logical origins" (Kahneman, 2003).

The economic value of ecosystems has two distinct aspects. One can be described as the sum of the ecosystem service benefits provided, referred to as final services (see Section 4) or output value. Another aspect considers the ecosystem's

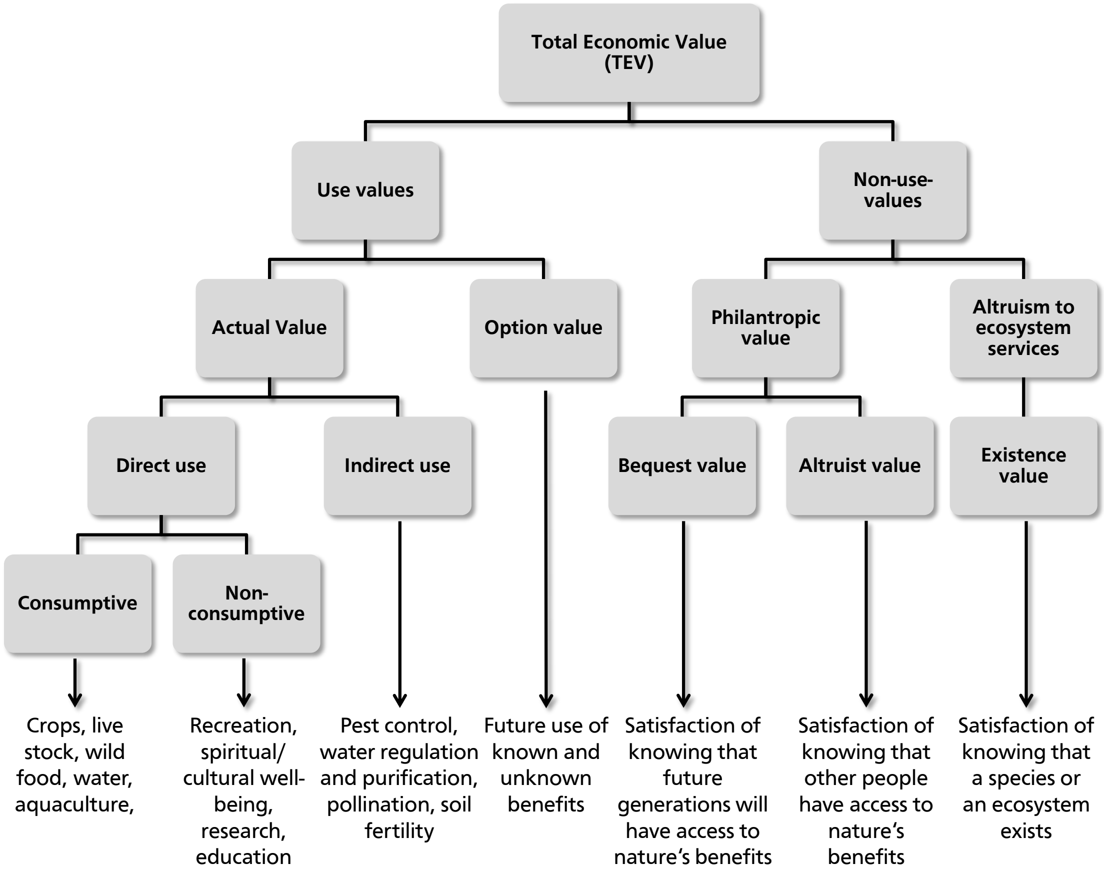

Figure 4.9: Illustration of the concept of Total Economic Value (TEV); based on TEEB (2010a)

capacity to maintain a sustained flow of these final services. This insurance value (Turner et al., 2003; Balmford et al., 2008) depends on the ecosystem's functioning based on ecosystem processes and structures (see Figure 4.8). The resilience (capacity to absorb disturbances) and the self-organizing capacity (to maintain essential functions) of an ecosystem is an important part of its insurance value (Holling, 1973; Walker et al., 2004). Biodiversity is strongly related to the insurance value of an ecosystem.

The (total) output value can be described using the Total Economic Value (TEV). The concept of TEV is the most prevalent assessment method for the economic value of ecosystem services (Pearce and Turner, 1990). It encompasses all values of service flows that natural capital generates at present as well as in the future by taking appropriate discounting rates into account. Ecosystem services, in theory, are valued for marginal changes in their provision. Hence, from an economic point of view ecosystems and their biodiversity can be considered as natural capital with ecosystem services as the interest that society receives from it (Costanza and Daly, 2006). The natural capital asset is scarce if the use, i.e. obtaining an additional unit of the service, of it implies opportunity costs, i.e. something else has to be given up. The basic assumption behind ecosystem service valuation is that society can somehow assign values to ecosystem services to the extent to which these fulfill human needs or result in direct or indirect satisfaction. While there are reasonable values for many provisioning services (e.g. wood or agricultural production) as there are often markets for these goods, the value of cultural or regulating services representing non-marketed goods is not clear. This and the public or common pool character of these services makes them a particular subject to overuse (Carpenter et al., 2006; Barbier, 2008). The TEV is used to express the sum of all components of utility (and disutility) that can be derived from ecosystem services expressed by using a common metric: money or any other (traded) exchange value that allows comparison. It is composed of use-values, direct, indirect and option value, and non-use values, bequest and existence value. Figure 4.9 illustrates the composition of the TEV and provides examples for different value types.

The principal logic behind valuation of ecosystem services is to illustrate the complexities of socio-ecological relationships, to make explicit how human actions affect ecosystem service values, and to express these values in units (e.g. monetary) that allow their incorporation in public decision-making processes (Mooney et al., 2005). Hence, quantifying the benefits from ecosystems through economic valuation of ecosystem services provides a means to assess the impacts and trade-offs of ecosystem change, especially when gains and losses are accrued by different beneficiaries at disparate spatial and temporal scales.

Valuation approach		Method	Value type
Market valuation	Price-based	Market prices	Direct and indirect use
	Cost-based	Avoided cost	Direct and indirect use
		Replacement cost	Direct and indirect use
		Mitigation, restoration cost	Direct and indirect use
	Production-based	Production function approach	Indirect use
		Factor income	Indirect use
Revealed preferences		Travel cost method	Direct (indirect) use
		Hedonic pricing	Direct and indirect use
Stated preferences		Contingent valuation	Use and non-use
		Choice modeling, conjoint analysis	Use and non-use
		Contingent ranking	Use and non-use
		Deliberative group valuation	Use and non-use

Table 4.7: Relationship between valuation methods and value types (TEEB, 2010a)

4.3.1 Economic valuation methods

Although a multitude of valuation approaches exist, two general valuation paradigms can be distinguished. One paradigm represent *biophysical methods* which derive values from measurements of the physical costs of producing a certain service based on a *cost of production* perspective (TEEB, 2010a). Thus, these methods consider the physical costs of labor, energy or material inputs, etc. of maintaining a desired ecological status in valuing ecosystem services. Another valuation paradigm are *preference-based methods* which, in contrast to biophysical methods, rely on models of human behavior assuming that values can be derived from the subjective preferences of individuals or groups.

There is an ongoing debate about the use of valuation methods, see Gomez-Baggethun and de Groot (2010). This dissertation deals primarily with cost-based and preference-based methods.

Values for ecosystem services in the context of TEV are derived from information of individual behavior which may be provided by market transactions directly related to the service in question. This valuation approach is referred to as direct market valuation. If price information of direct market transactions is absent, alternative price information can be derived from parallel market transactions, indirectly associated with the service to be valued. This second approach is termed revealed preferences approach. A third category of valuation techniques is represented by stated preferences approaches which in absence of direct and indirect price information on ecosystem services use hypothetical markets in order to elicit values (Chee, 2004). Table 4.7 illustrates the three principal valuation approaches, commonly used methods and the respective value they are used to assess.

Three main methods are used for direct market valuation: market price based methods, cost-based methods and methods based on production functions. Since all these methods use information from actual markets, they have the advantage of reflecting actual preferences or costs to individuals in a given market context. Additionally, prices, costs and quantities in existing markets are relatively easy to obtain, thus, may serve as a reference.

For the estimation of values for provisioning services *market price-based methods* are most often used. The goods produced on the basis of provisioning services are often sold on markets (e.g. crops on local food markets) and market prices, in well-functioning markets, reflect preferences and marginal cost of production. In this case these prices can be taken as accurate information on the value of these goods. In this case these prices can be taken as accurate information on the value of these goods. The product of the price of a good produced by the service and the marginal product of the ecosystem service can be used as an indicator for a provisioning ecosystem service value.

If ecosystem services benefits had to be created through artificial means the costs for doing so can be estimated using *cost-based methods* (Garrod and Willis, 1999). At least three techniques exist for cost-based methods: the costs that would incur in absence of an ecosystem service are expressed using the *avoided cost method*, the cost of replacing ecosystem services with artificial means is expressed using the *replacement cost method* and the cost of mitigating the effects of the loss of ecosystem services or restoring them is expressed using the *mitigation* or *restoration cost method*.

The contribution of an ecosystem service, e.g. a regulating service, to the delivery of another ecosystem service traded on a market is estimated using *production function-based approaches*. Thus, this approach considers the enhancement of income or productivity by a related ecosystem service. For the production function-based approach, in general, scientific knowledge on cause-effect relationships between the ecosystem service that is valued and the level of productivity or income increasing output is used. Thus, the physical effects of changes in an ecosystem service on an economic activity have to be determined first. Then, the impact of these changes is valued through corresponding changes in marketed outputs (Barbier et al., 1994).

Direct market valuation approaches have important limitations when applied to ecosystem services mainly due to the absence of or distorted markets for ecosystem services. If there are no markets for ecosystem service (or for goods or services indirectly related to them), then the information needed to carry out these approaches is also lacking. In case of existing markets these can be significantly distorted, for instance, by subsidies or limited competition. In this case prices will not adequately reflect neither preferences nor marginal costs which consequently leads to biased and unreliable values for ecosystem services. Specific problems are reported in using the replacement cost method under uncertainty (Barbier, 2007), with limited knowledge and information to define production functions to quantify cause-effect relationships (Daily et al., 2000; Spash, 2000), and with respect to the interconnectivity and interdependency of ecosystem services which may lead to double counting (Barbier et al., 1994; Daily, 1997).

In the absence of price information for ecosystem services their value may be derived indirectly from associated parallel market transactions using revealed preference approaches. Hence, these approaches are based on observed individual choices in existing markets that are related to the ecosystem service to be valued and which are supposed to *reveal* individual preferences indirectly. The most common methods for this approach are the Travel Cost Method (TCM) and Hedonic Pricing (HP). The TCM is mainly used to estimate recreational values related to ecosystem services and biodiversity. The rational for this method is that recreational experiences are associated with direct expenses and opportunity cost of time which relate to the quality and quantity of recreational sites. For the HP method information on an implicit demand for ecosystem services is utilized to value ecosystem services indirectly. It is often used to estimate economic benefits or costs associated with environmental quality, e.g. air or water pollution, or with environmental amenities, e.g. aesthetic views. Hence, the HP assumes that the price of a marketed good or service is related to its characteristics, or the services it provides. Revealed preference approaches, like direct market valuations, can be influenced by market imperfections which results in distorted values for ecosystem services, Moreover, these approaches often require large data sets and complex statistical analysis which makes them expensive and time consuming in operation. Their main drawbacks are the inability to estimate non-use values and the dependence on assumptions about the relationship of ecosystem service values and surrogate markets examined (TEEB, 2010a).

To simulate demand for use and non-use ecosystem services, stated preferences approaches are applied by using surveys on hypothetical (policy-induced) changes in the provision of ecosystem services. These approaches can be used independently of the existence of surrogate markets from which values of ecosystem services can be deduced. Three groups of techniques can be distinguished: Contingent Valuation (CV), Choice Modeling (CM) and group valuation.

The CV method uses questionnaires to find out about people's Willingness To Pay (WTP) for an increase or enhancement of ecosystem service provision, alternatively it can survey the Willingness To Accept (WTA) ecosystem service loss or degradation. CM is a method that attempts to model the decision making process of individuals in a given context by offering different alternatives (i.e. choices) with shared attributes to be valued, but on different levels (Hanley et al., 1998). The main difference between CV and CM is that CV methods usually present only one option to respondents. Respondents in a CM method are asked to consider all the options by trading off their different attributes. While CV is easier to design and implement, CM may provide estimates which are of more value for changes in the characteristics of ecosystems (TEEB, 2010a).

Another stated preferences approach is group valuation which combines stated preferences methods with techniques of deliberative processes from political sciences (Spash, 2008). These methods are used increasingly in order to take issues like value pluralism (e.g. public nature of ecosystem services and the measurement of their economic value through individual expression), incommensurability, non-human values and social justice into account (cf., Wilson and Howarth, 2002).

The limitations of stated preferences methods relate to a large extent to the issue of objective choice. Although they are often the only option to estimate non-use values, whether respondents' hypothetical answers correspond to their actual behavior in a real life situation remains doubtful. The literature on stated preference methods also reveals a general divergence between the results of WTP and WTA studies (Hanemann, 1991). Furthermore, depending on the ecosystem service to be valued there can be significant cognitive and knowledge constraints among respondents. Therefore, stated preference methods sometimes incorporate basic information upfront in questionnaires. According to Christie et al. (2012), valuation workshops that provide respondents with opportunities to discuss and reflect their preferences can help to overcome some of these constraints. In addition, deliberative methods were developed to integrated deliberative processes in environmental valuation as a response to criticism of contingent valuation (Sagoff, 1998). It is often the institutional setting that influences individual preferences in a valuation situation by activating particular motivations and rationalities (Vatn, 2005). According to Gowdy (2005), in cooperative consensus building and collective decision making, common preferences can result in shared values, i.e. institutions on perceived values. Deliberative valuation methods, e.g. facilitated as Participatory Rural Appraisals or Round tables, do not assume pre-existing values and emerge from a communicative social process (TEEB, 2010a). Understanding ecosystem valuation as a value-articulating institution (cf. Jacobs, 1997) in this way allows for expressing values, influences how values are formed and what kind of conclusions are to be drawn based on them.

Table 4.8 summarizes the different valuation methods and gives some examples of their application from literature (TEEB, 2010a).

Valuation method	Comment / Example	Reference
Market valuation		
Market prices	Mainly applicable to goods, only some cultural (e.g. recreation) and regulating services (e.g. pollination)	Brown et al. (1990); Kanazawa (1993)
Avoided cost	Derivation of flood control service from estimated flood damage	Gunawardena and Rowan (2005)
Replacement cost	Valuation of groundwater recharge based on the cost of an alternative source; cost of waste water treatment if wetlands are lost	Ammour et al. (2000)
Mitigation / restoration cost	Cost of preventive costs in absence of wetland serving as flood barriers	Breaux et al. (1995)
Production function/factor income	Improved soil fertility's impact on income of farmers, water quality improvements and increase of fish catch	Pattanayak and Kramer (2001)
Revealed preferences		
Travel Cost	Recreational value of ecosystems estimated through time and money people spend in traveling to the site	Martín-López et al. (2009)
Hedonic Pricing	Presence of water related to real estate prices	Garrod (1996)
Stated preferences		
Contingent Valuation	Estimation of non-use values, often based on survey questionnaire to identify peoples WTP (e.g. improved water quality)	Wilson and Carpenter (1999)
Choice modeling	Values are inferred from hypothetical choices or trade-offs that people make (e.g. finding of a landfill site)	Hanley et al. (1998)
Group valuation	Addresses shortcomings of revealed preferences methods (e.g. preference construction, lack of knowledge)	Spash (2008)

Table 4.8: Monetary valuation methods and examples of application from the literature

Provisioning services have been valued mainly through production function and direct market valuation approaches, while for regulating service values avoided cost, replacement and restoration costs, or contingent valuation have been used mostly (Martín-López et al., 2009). Hedonic pricing for aesthetic information, contingent valuation for existence values and travel cost methods for valuation related to recreation and tourism have instead been used mainly for cultural services. Often there are specific valuation methods used according to certain ecosystems (TEEB, 2010a).

Since the 1980s several authors have developed ecosystem service valuation methodologies, among others, Dixon and Hufschmidt (1986); Pearce and Turner (1990); Freeman (1993); Hanley and Spash (1993). Authors such as Kramer et al. (1995) and van Beukering et al. (2003) carried out valuations for particular ecosystems like forests or coral reefs. Furthermore, several frameworks for the valuation of ecosystem services have resulted from broad and far reaching studies (Costanza et al., 1997; Turner et al., 2000; de Groot et al., 2002; MEA, 2003). Especially the publications of Costanza et al. (1997); MEA (2005); Stern (2006); TEEB (2009) brought the importance of valuing ecosystem services to the attention of policy makers.

Figure 4.10 illustrates the relationship between elements of an ecosystem (structure, process, functions and the provision of goods and services), on the one hand, and the aggregation of (ecological, socio-cultural as well as economic) values to a total value as basis for a decision-making process to determine policy options and management measures, on the other hand. These outcomes of the decision-making process will in turn have an impact on ecosystem structures and processes. The grey shaded elements in Figure 4.10 are determined by the human system, while the white elements form the natural system. Ecosystem goods and services represent the interface of the natural and human system for ecological, socio-cultural and economic valuation. However, ecosystems themselves, i.e. their structures and processes, are of ecological value as well since they are essential for the functioning of ecosystems, independent of the provision of ecosystem goods and services.

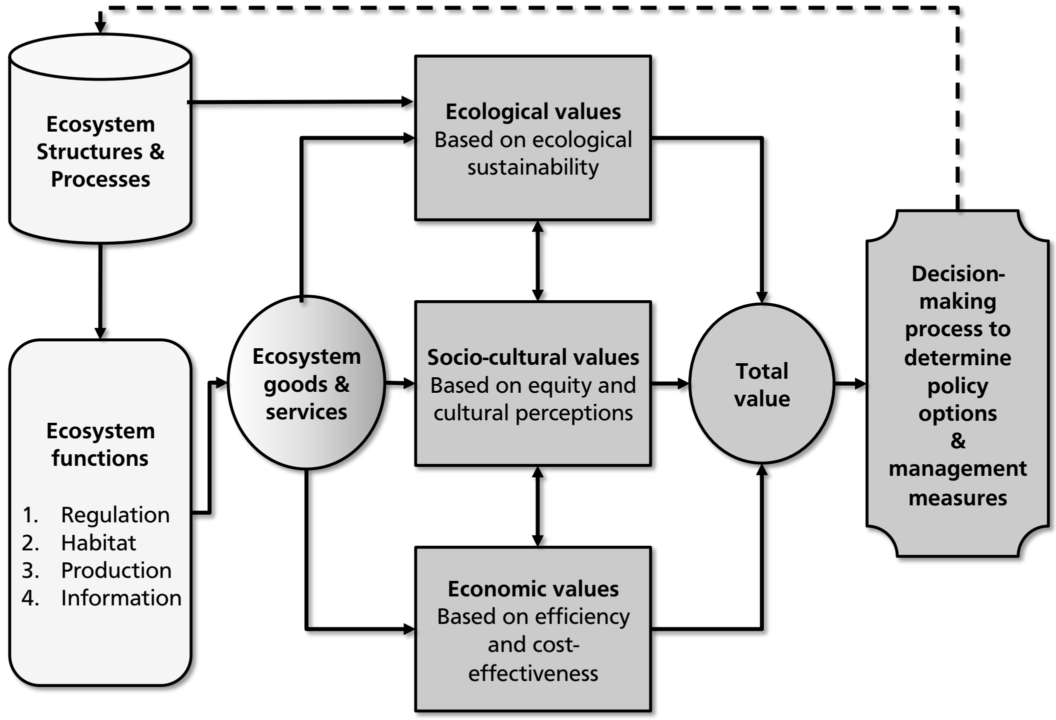

Figure 4.10: Process of ecosystem valuation; based on de Groot et al. (2002)

According to Hein et al. (2006), the valuation process of ecosystem services in a policy context consists of five principal steps:

1. Specification of the boundaries of the ecosystem to be valued
2. Assessment of the ecosystem services supplied by the system
3. Valuation of the ecosystem services
4. Aggregation or comparison of the values of the services
5. Analysis of scales and stakeholders

Especially the last step of the valuation process stresses the need to explicitly consider the scales at which ecosystem services accrue to the different stakeholders. Since ecosystem services are often generated and provided at particular scales, valuation requires assessing at which scale, and to whom the benefits of the system's services accrue. The assessment of generating and provisioning scales as well as of the stakeholders involved is critical to find an appropriate management scale defined by fit and interplay. The identification of scales and stakeholders, according to Hein et al. (2006), "allows the analysis of potential conflicts in environmental management, in particular between local stakeholders and stakeholders at larger scales". The maintenance of hydrological services provided by a forest in an upper watershed, for instance, implies restrictions on the use of the forest by local stakeholders. Thus, a basis to determine the size of potential compensation payments to local users may be provided through the analysis of different (opportunity) costs and benefits of ecosystem management for stakeholders at different scales (Hein et al., 2006). The definition of the spatial, temporal and socio-economic scale is a critical step in an economic valuation as benefits can be perceived far away in time and space from the ecosystem that provides them. The spatial scale establishes service specific provider-beneficiary relationships, while the temporal scale influences when benefits are actually perceived and may require discounting (cf., Korsgaard and Schou, 2010).

Economic valuation of ecosystem services may also bear significant downsides. The assumptions underlying conventional economic valuation are marginality and substitutability. According to Korsgaard and Schou (2010), both assumptions are critical for valuing ecosystem services, especially in developing countries. Marginality means that for conceptual and practical reasons the change to be valued must be marginal. In practice, for instance, there is not always a straightforward relationship between impact and resulting change, making it difficult to judge the marginality of a change (Limburg et al., 2002). The assumption of substitutability in utilitarian valuation approaches may also be critical since it implies that all types of values and capital are substitutable or replaceable. In cases where ecosystem services are used for subsistence, for example rural people in developing countries who depend directly on them, this assumption may not be valid (Korsgaard and Schou, 2010). But this does not only apply to subsistence use of ecosystem services, there is general recognition that substitution of natural capital and ecosystem services has limits and a critical amount of natural capital has to be preserved

(Barbier et al., 1994; Daly and Townsend, 1996; Prugh et al., 1999; Daly and Farley, 2004). Moreover, substitution of ecosystem services is often only partial for some desired ecosystem services. For example, water treatment plants can substitute for ecosystems in providing clean drinking water but this will not overcome the impacts of water pollution on other components of the ecosystem and the services they provide. And if these substitutions benefit other beneficiaries than the previous beneficiaries of these ecosystem services, the substitution has additional distributional effects. Hence, in order to take the non-marketed value of ecosystem services into account, economic valuation, as one way of value expression, should be embedded into a superior value setting of ethical as well as moral values of nature to society (Sagoff, 1988). However, the intrinsic value of nature for instance cannot be reflected by prices and economic valuation. Valuing of existing ecosystems, landscapes, certain species or other elements of biodiversity is recognized by all human societies and a feature of all cultures and sometimes cultural or spiritual values alone are already sufficient to ensure sustainable use and conservation of ecosystems (TEEB, 2010b). In this context the irreversibility of ecosystem or biodiversity loss is also a critical issue. It is not possible to value an irreversible loss. The formulation of safe minimum standards is an approach to deal with problems of uncertainty and irreversibility (Costanza et al., 2001).

Since ecosystems as well as all the functions and services they provide are not entirely substitutable, prices for ecosystem services can only express what actually is substitutable, e.g. the way a service is provided as the value of actions taken in order to assure a continuous provision or improvements in Ecosystem Services supply. Therefore, valuation can be a way of organizing information to help guide decisions but is not a solution or end in itself (Daily et al., 2000). In this regard, Farley (2010) stresses that there is also much to be gained from examining the problem through an economics lens. The author points out that through the recognition of economic values of ecosystems and the services they provide, environmental protection which is also driven by economic factors may be easier to understand and ecosystems may be conserved better by reducing degradation. Moreover, economics can offer a when-to-stop, i.e. when-to-conserve rule is another point which Farley (2010) stresses. Thus, if economic systems grow by displacing and degrading ecosystems, then marginal benefits of economic growth are diminishing while the marginal costs of ecological degradation are increasing. Hence, the point to stop economic growth and to focus stronger on ecological conservation is reached when the two are equal. Finally, economics may help to understand how to efficiently and justly allocate resources toward conservation (Farley, 2010).

4.3.2 Economic valuation of hydrological ecosystem services

Economic valuation of hydrological ecosystem services has been made since at least the 1970s (Wilson and Carpenter, 1999; Echavarría, 2000). The economic quantification of environmental amenities such as water quality has been the principle objective of these early valuation studies (Leggett and Bockstael, 2000; Carson et al., 2003). However, as Brauman et al. (2007) point out, the value of actual ecosystems to the production of these valued amenities is less clear. Often a proxy related to the production of an attribute such as water quality is used in valuation studies of wetlands and riparian buffers, which then value the incremental impact of changes in that attribute (Pattanayak, 2004). This way, the gains in improved water quality and reduction in soil loss from vegetated buffer strips were proven to outweighed the costs of land taken out of production in a study of Californian farmers (Rein, 1999). Different values were estimated for wetlands in a study of flood damages related to wetland development in Florida (Highfield and Brody, 2006). Wetlands within floodplains received higher values than those outside of flood-prone areas. Furthermore, different approaches have been undertaken in order to estimate the value of different hydrological services produced by river basins (Brauman et al., 2007). Kaiser and Roumasset (2002) have modeled the role of conserved forests in Hawaii in the context of groundwater recharge. The authors' results indicate that in some places the value of increased groundwater recharge under conserved forest is substantial; the calculation for one forested river basin is substantial, the calculated of one forested river basin was estimated to be between US-$ 1.5 and US-$ 2.5 billion (Kaiser and Roumasset, 2002). With regard to the effect on the timing of water for in-situ and diverted supply, the value of forests may also be high. A study of Guo et al. (2000) in China indicates that forested river basins may improve the productivity of dams on the Yangtze river and increase revenues by up to US-$ 600,000 per year. The economic value of native forest in a river basin in Chile was estimated to be over US-$200 per hectare and year because of its positive effects on drinking water supply (Núñez et al., 2006). However, the functionality and resulting values of ecosystems providing hydrological services is very site-specific and likely to be highly variable, despite the striking results of valuation studies (Woodward and Wui, 2001).

In a practical management and policy context, the value of hydrological ecosystem services has often to be assessed in terms of comparisons of service production from an alternative land use. Chomitz et al. (1999), for instance, highlight that tropical forest conservation may provide net hydrological benefits compared to an alternative land use of annual cropping or grazing, but not when compared agroforestry. Besides the choice of a comparative land use, value is also dependent on the scale at which services are assessed. A study of drinking water treatment costs in relation to the relative share of forested area within the river basin in the US revealed the importance of the spatial extension of forests for the provision of hydrological services (see Table 4.9).

Share of watershed forested	Treatment costs per 3,785 m^3	Average annual treatment costs	Cost increase over 60 % forest cover
60%	US-$ 37	US-$ 297,110	-
50%	US-$ 46	US-$ 369,380	24 %
40%	US-$ 58	US-$ 465,740	57 %
30%	US-$ 73	US-$ 586,190	97 %
20%	US-$ 93	US-$ 746,790	151 %
10%	US-$ 115	US-$ 923,450	211 %

Table 4.9: Forest cover and related water treatment costs based on 27 US water supply systems[1]; from Postel and Thompson (2005) based on Ernst (2004)

[1] Based on treatment of 22 million gallons (83,270 m^3) per day, the average daily production of the water suppliers surveyed.

Investments in watershed protection for the improvement of metropolitan water supplies and the reduction of water treatment costs are particularly popular in the US. Postel and Thompson (2005), in their review, document avoided costs of several major US cities which have invested in watershed protection instead of constructing (additional) drinking water filtration plants (see Table 4.10). The case of New York City is one of the most cited examples of cost-effective investment in the provision of hydrological ecosystem services compared to built-infrastructure and has served as a prototype for many replications (NRC, 2000; Appleton, 2002; Pires, 2004).

Metropolitan area	Population (thousands)	Avoided costs through watershed protection [US-$]	Source
New York City[1]	9,000	1.5 billion spent on watershed protection over 10 years to avoid 6 billion in capital and 300 million in annual operating costs.	NRC (2000)
Boston, Massachusetts	2,300	180 million (gross) avoided cost.	Stearns (2000)
Seattle, Washington[2]	1,300	150-200 million (gross) avoided cost.	Flagor (2003)
Portland, Oregon	825	920,000 spent annually to protect watershed is avoiding a 200 million capital cost.	Reid (2001)
Portland, Maine	160	729,000 spent annually to protect watershed has avoided 25 million in capital costs and 725,000 in operating costs.	ECONorthwest (2004)
Syracuse, New York	150	10 million watershed plan is avoiding 45-60 million in capital costs.	ECONorthwest (2004)
Auburn, Maine	23	570,000 spent to acquire watershed land is avoiding 30 million capital cost and 750,000 in annual operating costs.	Ernst (2004)

Table 4.10: US cities avoiding construction of filtration plants through watershed protection (Postel and Thompson, 2005)

[1] The City is being required to construct a $687 million filtration plant for the more-developed Croton watershed, which supplies about 10% of the city's water. The filtration waiver applies to the Catskills-Delaware watershed, which supplies about 90% of the city's water.
[2] Supply from Seattle's Cedar River watershed is unfiltered, but that from the Tolt watershed is now filtered.

Besides cost-based economic valuations, preference-based valuations have also been carried out. According to Wilson and Carpenter (1999) the three most prevalent methods to value non-marketed hydrological ecosystem services applied in the US are the Travel Cost Method (TCM), Hedonic Pricing (HP), Contingent Valuation (CV), or combinations of them.

There are also valuation studies of hydrological ecosystem services for developing countries. Korsgaard and Schou (2010) reviewed several recent studies and compared the valued hydrological services, the valuation method as well as the estimated values (see Table 4.11). The valuation methods most frequently applied are based on replacement, mitigation, or avoided cost estimates, or a combination of them. Price-based market valuation was applied for direct water use only. In order to estimate recreational values or landscape beauty travel cost based and contingent valuation methods were used. The review illustrates the large variety of economic values assigned to different hydrological services. The range is especially wide in the case of water purification and flood mitigation, but also for the other services there is a considerable difference in values. These differences are inherently method and context specific. Woodward and Wui (2001), for instance, based on a comprehensive meta-analysis of 39 studies of economic values of wetland services conclude that value estimations are very site specific and point out that the use of benefits transfer to estimate wetland values faces substantial challenges. Thus, the need for site-specific studies remains. Most studies reported their results in the unit of net value per hectare and year. This unit is suitable if the value of an ecosystem service is correlated to the size of its provisioning area. However, for some services such as recreational ones this unit is less useful (cf. Korsgaard and Schou, 2010). Moreover, apart from being possibly related to the size of the provisioning, the value of hydrological services is determined by the characteristics of the beneficiaries. The value of flood mitigation, for instance, is highly dependent on a given context, e.g. the size of the affected population and potential damages. As the majority of studies reviewed by Korsgaard and Schou (2010) provide information on the affected population, the authors converted the values of the per hectare values into per capita values (see Table 4.11). According to the authors, this unit results in more consistent values for most services with total values within the range from 10 to 230 US-$ / capita / year. This demonstrates that valuation can reflect both the value of the provisioning ecosystem and the value perceived by beneficiaries. Therefore, Korsgaard and Schou (2010) argue "that the potential value of an ecosystem service is a function of ecosystem (or biophysical) characteristics (e.g. size) while the actual value (the extent to which the potential value is utilized) is a function of population (or socioeconomic) characteristics".

Hydrological service	Valuation method	Net economic value	
		[US-$ / ha / year]	[US-$ / per cap / yr]
Direct water use	Direct price-based market valuation	50, 150, 400[1]	1, 10, 21[2]
Water purification	Replacement or mitigation cost	20, 40, 140, 620, 1400[3]	2, 8, 20, 50[4]
Flood mitigation	Replacement, mitigation or avoided cost	2, 30, 90, 340, 1400, 1700[5]	2, 2, 20, 75, 370[6]
Groundwater recharge	Replacement, mitigation or avoided cost	10, 70, 90[7]	25, 30[8]
Erosion control	Replacement, mitigation or avoided cost	20, 120[9]	7[10]
Recreation, tourism	Travel cost, contingent valuation	20, 30, 260[11]	20, 1100[12]

Table 4.11: Economic values from studies in developing countries for different hydrological ecosystem services; based on (Korsgaard and Schou, 2010)

[1] Acharya (2000); Emerton (1994); Seidl and Moraes (2000)
[2] Emerton and Kekulandala (2002); Emerton (1994); Acharya (2000)
[3] Turpie et al. (1999); Gerrard (2004); Seidl and Moraes (2000); Emerton and Kekulandala (2002); Emerton et al. (1999)
[4] Gerrard (2004); Emerton et al. (1999); Emerton and Kekulandala (2002); Turpie et al. (1999)
[5] Turpie et al. (1999); Emerton et al. (2002); Rosales et al. (2003); Seidl and Moraes (2000); Gerrard (2004); Emerton and Kekulandala (2002)
[6] Turpie et al. (1999); Emerton et al. (2002); Emerton and Kekulandala (2002); Gerrard (2004); Rosales et al. (2003)
[7] Turpie et al. (1999); Seidl and Moraes (2000); Acharya (2000)
[8] Acharya (2000); Turpie et al. (1999)
[9] Seidl and Moraes (2000); Emerton et al. (2002)
[10] Emerton et al. (2002)
[11] Emerton and Kekulandala (2002); Busk (2002); Seidl and Moraes (2000)
[12] Emerton and Kekulandala (2002); Busk (2002)

When looking at the value of hydrological ecosystem services to different beneficiaries it becomes clear that it depends strongly on the purpose of the water use. For instance, hydrological services in form of provisioning services for irrigated agriculture are usually valued the lowest, although this depends on the value of the crop. Provisioning services used for high-value crops can be much higher in value, sometimes even in the order of magnitude to the value of water in domestic and industrial uses. However, if hydrological services are used to supply domestic water its value is generally highest, whereas the values for environmental purposes, e.g. environmental flows, maintenance of wetlands, wildlife habitat, vary widely, but typically fall in between the agricultural and domestic values (Brouwer, 2010).

Prices for the use of hydrological ecosystem services exist, in most cases, for marketed provisioning services only. However, river basins provide several other non-marketed hydrological services such as the storm protection functions provided by mangroves or the water filtration functions provided by wetlands (Gleick, 1993; Naiman et al., 1995; Daily, 1997). Goldberg (2007), for instance, argues that although various studies recognize that degradation of hydrological ecosystem services "[...] represents a loss of capital assets, significant gaps remain at the policy and methodological levels in terms of economically quantifying both the costs of water related investments as well as the direct and indirect costs of watershed degradation, and the multiple benefits of supporting integrated water resource management". Moreover, Krchnak (2007) concludes in her review of current river basin management practices employed across the Western Hemisphere that a vast majority of them does not fully capture the non-marketed economic values of river basins.

Goldberg (2007) stresses that accounting for non-market values of river basins and integrating upstream variables into current management practices, "[...] provides leverage for an alternative management option in which policymakers can manage water resources more holistically and in which the livelihoods of upstream communities may be simultaneously improved (e.g., farmer's enhanced long-term return via soil conservation). Thus, internalizing the non-market values (externalities) tied to watersheds will result in more socially and economically optimal use of water systems". This approach, contrary to traditional downstream oriented approaches, can stimulate due consideration toward the inextricable link between beneficiaries and hydrological functions of river basins.

Without efforts to quantify the non-market benefits associated with goods and services generated by watersheds, policy and managerial decisions hold the potential to be skewed in favor of environmentally degrading practices by neglecting the diffuse interests that benefit from many of the non-market oriented characteristics of such systems (Wilson and Carpenter, 1999). To curtail the impact of such negative externalities and correct the total economic value of environmental assets, economists and other policy-oriented social scientists have developed a diverse array of techniques to measure the value of non-market environmental goods and services.

4.4 Addressing the governance challenge of fit and interplay in the context of IWRM implementation

In the freshwater chapter of Agenda 21, issued in 1992 at the UN Conference on Environment and Development (UNCED), the governments defined IWRM as a process based on water being "an integral part of the ecosystem, a natural resource and a social and economic good, whose quantity and quality determine the nature of its utilization". This more than 20-year-old agreement on a definition for IWRM highlights the importance of ecosystems and water being an integral part of them.

In the IWRM process of the last two decades, ecological approaches prevailed more and more. The control of reservoir discharge according to environmental flows or the recovery of natural floodplains as retention storage in case of floods are vital examples. Both examples illustrate that natural flow regimes are gaining importance in the context of IWRM although society has to accept (substantial) trade-offs, e.g. the loss of construction land on floodplains. Another example is the use of (constructed) wetlands for waste water treatment. With the establishment of the European Water Framework Directive in 2000 and the aim to achieve a good ecological status of water bodies, the ecosystem approach to water resources management was further strengthened, at least in Europe.

In the IWRM context, an ecosystem approach to water management is achieved through the management of water and land resources in an integrated manner on the basis of river basins as Integrated River Basin Management (see Section 3.1.1). According to Reynolds (1993), who summarizes the conceptual basis for ecosystemic water management based on contributions to the United Nations Economic Commission for Europe (UNECE) seminar on Ecosystems Approach to Water Management held in 1991, "the ecosystemic [or ecosystem] approach is proactive in seeking to defend ecological integrity per se and to devise action appropriate to improving ecosystem condition. In essence, it becomes paramount to determine the current state of the ecosystem in question, and the processes currently dominating its behaviour, and to implement action to enhance its future functioning". Thus, essential importance is placed on the ecosystem as integrator for water management. This implies the recognition of ecosystems as equivalent water users or co-users (cf., Reynolds, 1993) as well as the integration of water and land as management subjects. The benefits of the ecosystem approach to human well-being is also made explicit: "Benefits accrue from new uses of revitalized ecosystems, supporting new livelihoods from harmonious subsistence or the commercial use of restored catchment areas, more enjoyable leisure and recreation, and more natural flood control and nutrient removal by vegetation" (Reynolds, 1993).

However, in the context of IWRM, the ecosystem approach has been interpreted mainly in form of environmental flow requirements, i.e. water for ecosystems (cf. Forslund et al., 2009), rather than ecosystems as service providers like in the ecosystem services concept. Thus, so far in IWRM an ecosystem approach has been applied principally in an ecological sense alone. Some authors consider the management of water resources on the basis of river basins as management unit already an ecosystem approach, others refer to an ecosystem approach in the context of environmental flows and incorporation of the natural ecosystem as equivalent water "user" (e.g. UNCEC, 1993; Jewitt, 2002; Hooper, 2003; Leendertse et al., 2008). The use of an ecosystem service approach that connects water users and the natural ecosystem directly has not found equal recognition.

In contrast, the ecosystem service approach, although being essentially based on the ecosystem approach, goes further in making the *link* between ecosystems as service providers and society as ecosystem service beneficiary explicit. The ecosystem service concept, first and foremost, serves as a communication tool which increases the visibility and recognition of the importance of ecosystems for human well-being beyond their role as a resource stock (cf., Daily, 1997). A valuation in economic terms, then, is making ecosystem services visible on the balance sheet. Thus, the ecosystem service concept provides incentives, e.g. persuasive or economic, to change human behavior toward sustainable water management.

Furthermore, recognizing ecosystems as service providers can facilitate green infrastructure solutions, e.g. flood retention by natural floodplains or water purification in wetlands which may at least complement traditional built infrastructure. Moreover, the ecosystem service approach may not be limited to hydrological ecosystem services alone. When considering synergies among different ecosystem services there is also the potential to address other services, for

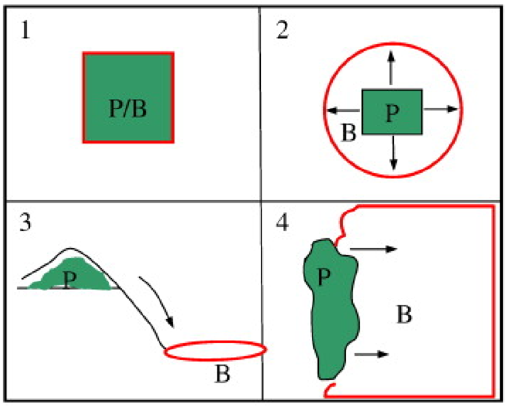

Figure 4.11: Possible spatial relationships between ecosystem service production (P) and benefit areas (B) (Fisher et al., 2009)

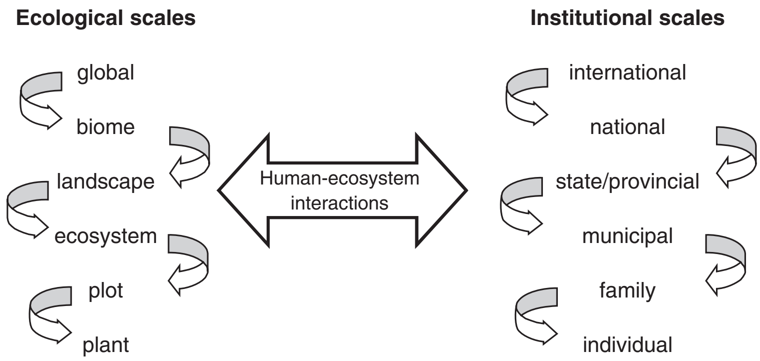

Figure 4.12: Selected ecological and institutional scales of human-ecosystem interaction; adapted from Leemans (2000) as presented in Hein et al. (2006)

instance pollination or carbon sequestration, important to specific stakeholders enabling broader cross-sectoral cooperation. Hence, a “step outside of the water box“ seems to become easier. As well as the ecosystem approach, the ecosystem service approach is based on understanding and learning about the properties of ecosystems but additionally focuses on the impact of human actions on the provision of services and humans as their potential beneficiaries.

Besides a better communication of the value of ecosystems and the broadening of the solution space by adding green infrastructure solutions, the ecosystem service approach establishes new cross-sectoral relationships between providers and off-site beneficiaries. This has important implications since providers and beneficiaries may now be identified in a spatially (and possibly as well temporally) manner, thereby creating explicit spatial scales for management. The cause-effect analysis at the core of the ecosystem service concept acknowledges the substantial importance of intact ecosystems for the provision of services that affect human well-being. However, it is a certain ecosystem state that provides the desired services, and humans (i.e. in this case land users) may facilitate the service provision through certain land use practices or conservation efforts. In the context of water resources management this represents the desired link between water and related land resources within a river basin through a direct involvement of respective stakeholders. This way, solutions to the problem of fit are not limited to ecosystem properties alone but include provisioning and use characteristics of ecosystems and their services as well. Hence, addressing institutional fit and interplay is inherent to the ecosystem service concept. Therefore, the identification of appropriate management units based on an ecosystem service approach can lead to much more context-specific bottom-up results than approaches that are based on natural (hydrological) boundaries versus political boundaries and basically top-down alone.

The relationship between the provision of ecosystem services and the benefits derived from them represents the core of the context-specific solutions of fit and interplay provided by the ecosystem service approach. In order to allow a more comprehensive understanding of the value and perceptions of relevant stakeholders, an assessment of ecosystem services has to take place both at ecologically understandable and policy-relevant scales (Hein et al., 2006; Fisher et al., 2008; Granek et al., 2010). Spatial explicitness in the context of ecosystem services is important as the provision and use or benefits of services vary spatially. The benefits from ecosystem services can range from being on-site at the point of provision to off-site locations at a local, regional or even global scale. Fisher et al. (2009) describe possible spatial relationships between service production areas (P) and service benefit areas (B); see Figure 4.11. The authors identify four possible spatial relationships: area of service provision and area of benefits are the same; e.g. soil formation, provision of raw materials, illustrated in (1) in Figure 4.11. An omni-directional provision with benefits at the surrounding landscape, e.g. pollination, carbon sequestration, as illustrated in (2). Some services have specific directional benefits (3 and 4), e.g. an uphill-downhill relationship (3), for instance, in water regulation services provided by forested slopes. And as illustrated in (4) service provision units could be coastal wetlands providing storm and flood protection to a coastline (Fisher et al., 2009). Scale and directional characteristics of ecosystem service provision should be considered together, e.g. local omni-directional service provision of pollination or regional directional service provision of flood protection. Thus, understanding the distribution of services and benefits across a landscape is important to effectively match providers and beneficiaries (Naidoo et al., 2006; Chan et al., 2006).

According to Fisher et al. (2009), applying this consideration allows a benefits-directed recognition of the spatial-temporal dynamics of ecosystems, public-private good aspects, and benefit dependence of services. Hence, an internalization of external effects and the development of additional cooperative structures is facilitated.

Analyzing scales of ecosystem service production and, even more importantly, the scales at which ecosystem services potentially contribute to benefits, is important in order to reveal interactions of different stakeholders and institutional interplay. This analysis allows connecting ecosystem service providers and beneficiaries in a spatially explicit manner and may provide further insight for the development of appropriate institutional scales for decision making in environmental management (Hein et al., 2006). The temporal and spatial scales at which ecosystem services are provided to the human, i.e. economic system varies greatly from short-term, on-site level (e.g., amenity services) to a long-term, off-site global level (e.g., carbon sequestration) (Turner et al., 2000; Limburg et al., 2002). The spatial scale at which ecosystem services are provided determines the stakeholders that may benefit from them, thus, provisioning scales and stakeholders correlate (Vermeulen and Koziell, 2002). The ecosystem processes and structural characteristics that influence ecosystem functioning are found over different spatial (and temporal) scales, ranging from plot-level plant-soil interactions, pests or fires at the meso-scale to geological and macro-climate process at the global level sometimes occurring at temporal scales of millenniums. According to Limburg et al. (2002), the aforementioned large-scale and long-period phenomena general set the physical constraints on the processes at smaller scales and shorter periods. These *exogenous* drivers of ecosystem service provision are hardly manageable, although, management of ecosystems can focus on increasing systems resilience and, hence, reduce the risk of undesired impacts. The joint impact of small-scale processes, however, can also drive large-scale processes, as the ecologist Levin (1992) found out. The cumulative activity of microbes operating on the scale of micrometers, for instance, steers the large scale process of nutrient cycling by nitrogen fixation and demineralization of organic material (Hein et al., 2006). Related ecosystem services may be perceived in turn at very different scales, e.g. enhancement of soil fertility through nitrogen fixation at the plot scale or carbon sequestration for global climate regulation. Variations in ecosystem service provision at different temporal scales are discussed by Howarth and Norgaard (1993) (discount rates and inter-generational equity) and Hanley and Spash (1993).

Just as different institutional scales for decisions on the utilization of natural resources in socio-economic systems can be distinguished (North, 1990; Becker and Ostrom, 1995), a distinction is equally possible for the ecological scale at which ecosystem services are provided (see Figure 4.12). Ecosystem services contribute to benefits at all institutional scales (Limburg et al., 2002), thus, reaching different stakeholders at these levels, ranging from individual households to local and internationally operating firms (Berkes and Folke, 1998; Peters and Meybeck, 2000). Therefore, the problem of fit (see Section 3.1) where ecological and institutional boundaries do not coincide is directly addressed in the concept of ecosystem services whose stakeholders (ecosystem service providers and beneficiaries) often cut across a range of institutional scales, e.g. administrative boundaries, and sectors (Cash and Moser, 2000; Hein et al., 2006). This way the ecosystem service concept inherently addresses the problem of fit.

The spatial scales at which ecosystem services are received differ between ecosystem service categories. While provisioning services often coincide with their production areas and sometimes off-site, the benefits from regulating services are generally perceived off-site from the production area. The benefits from provisioning services are defined on-site by the producing ecosystem, e.g. a lake for fish or a forest for wood. However, for regulation services which are typically generated at a specific ecological scale, the benefits may accrue to stakeholders at a range of institutional scales (see Table 4.12).

Ecological scale	**Dimensions (km^2)**	**Regulation services**
Global	> 1.000.000	Carbon sequestration Climate regulation through regulation of albedo, temperature and rainfall patterns
Biome - landscape	10.000 - 1.000.000	Regulation of the timing and volume of river and ground water flows Protection against floods by coastal or riparian ecosystems Regulation of erosion and sedimentation Regulation of species reproduction (nursery service)
Ecosystem	1 - 10.000	Breakdown of excess nutrients and pollution Pollination (for most plants) Regulation of pests and pathogens Protection against storms
Plot - plant	< 1	Protection against noise and dust Control of run-off Biological nitrogen fixation

Table 4.12: Ecological scales most relevant for the regulation of ecosystem services (some services may be relevant at more than one scale); from Hein et al. (2006) based on Hufschmidt et al. (1983); de Groot (1992); Kramer et al. (1995); van Beukering et al. (2003)

Moreover, besides a specific ecological scale, the position in the landscape also plays a role for the provider-beneficiary constellation. Hein et al. (2006), for instance, stress that the impact of the water buffering capacity of forests will be noticed only downstream in the same river basin (Bosch and Hewlett, 1982). Hence, affected stakeholders of regulating services are all residing in or depending upon the area affected by the service.

Table 4.13 groups the ecosystem services listed in Costanza et al. (1997) into five categories according to their spatial characteristics. Services like carbon sequestration are classified as global and non-proximal since the spatial location of carbon sequestration does not matter to global beneficiaries. Local proximal services, in turn, are characterized by their spatial proximity of the provisioning ecosystem to the beneficiaries. Costanza (2008) provides the example of storm protection which requires that the ecosystem providing the protection is proximal to the infrastructures being protected. The hydrological services of water supply and water regulation represent directional flow related services being dependent on the flow from upstream to downstream (Costanza, 2008). In the context of IWRM, spatial relationship of provision and benefit of local-proximal, directional flow-related and in-situ are of specific importance for the identification of appropriate units for management.

Spatial relationship of provision and benefit	Ecosystem Services
Global - non-proximal	Climate regulation
(Independent on proximity)	Carbon sequestration and storage
	Cultural / existence value
Local - proximal	Disturbance regulation / storm protection
(Depending on proximity)	Waste treatment
	Pollination, biological control, habitat / refugio
Directional flow-related	Water regulation / flood protection, water supply
(Flow from point of production to point of use)	Sediment regulation / erosion control
	Nutrient regulation
In situ	Soil formation
(Point of use)	Food production / non-timber forest products
	Raw materials
User movement related	Genetic resources
(Flow of people to unique natural features)	Recreation potential, cultural / aesthetic

Table 4.13: Ecosystem services classified according to their spatial characteristics (Costanza, 2008)

The heterogeneity and dynamics of ecosystem service provision and ecosystem service benefits describe potential cross-sectoral (e.g. agriculture and energy production or forestry and water supply or flood protection) as well as cross-scale ecosystem service provider and beneficiary constellations. The notion that ecosystems provide bundles of ecosystems implies that enhancement or management of one ecosystem service may have an impact on the provision of others. These ecosystem services interrelationships have to be considered in order to achieve the desired management goals. Bennett et al. (2009) analyzed the different ecosystem service relationships and propose a classification based on two principal mechanisms causing them: (1) effects of drivers on multiple ecosystem services (i.e. common drivers) and (2) interactions among ecosystem services (see x- and y-axis in Figure 4.13). Drivers of changes in ecosystem services, for instance fertilizer use or land use changes, can affect just a certain ecosystem service, thus, representing an independent driver, or they can influence several ecosystems at a time in form of a shared driver (shown along the x-axis of Figure 4.13 respectively). Despite the mechanism of sharing a common driver for ecosystem service change, direct interaction among ecosystem services is also common (shown along the y-axis of Figure 4.13). These mechanisms can be of a weak or rather a strong nature. Moreover, these mechanisms can have a positive or a negative effect on ecosystem services (see Figure 4.13; black arrows indicate a positive effect and gray arrows a negative effect). For instance, using fertilizer can improve crop production (see black arrow in sector 2 of Figure 4.13) while having the unintended effect of diminishing the provision of clean water (gray arrow). The provision of individual ecosystem services can be altered through the interaction between ecosystem services even if an ecosystem service is independent of a certain driver. Sector 3 of Figure 4.13, for example, illustrates how conservation tillage may have a positive (i.e. reducing) effect on soil erosion as it enhances the service of crop production which in turn reduces soil erosion. Hence, enhancement of one service by a driver can lead to synergies if multiple services respond positively to the enhanced service (Hey, 2002; Zedler, 2003).

Knowledge of the mechanisms of independent or shared drivers of ecosystem change and ecosystem service interactions can improve the management of trade-offs and synergies between services. In cases where ecosystem services have weak direct interaction it may be useful to address a shared driver if it has the same directional (positive or negative) effect on the considered ecosystem services. In contrast, addressing a shared driver may be unlikely to change trade-off in the long-term on ecosystem services if the services themselves have an enhancing interaction (Bennett et al., 2009). Therefore, Bennett et al. (2009) make the following three propositions for the management of ecosystem service relationships:

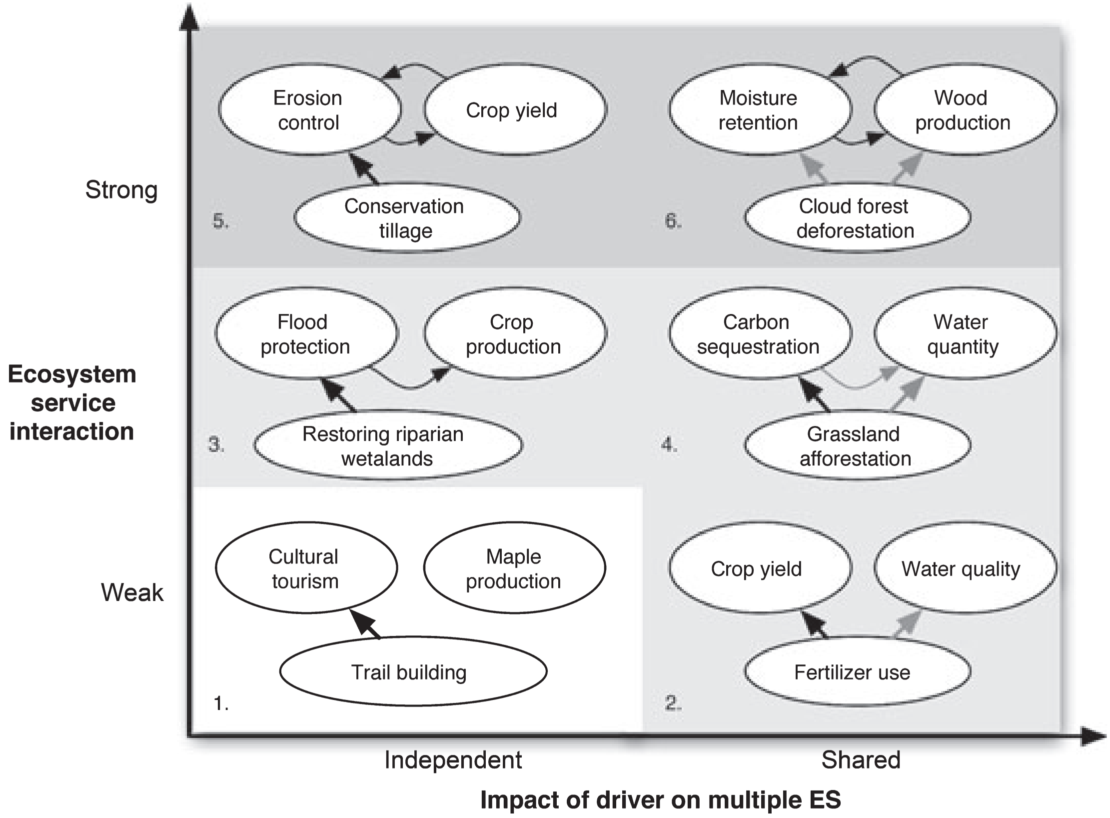

Figure 4.13: Supply of ecosystems services related either due to interactions between ecosystem services or due to responding to the same driver of change (Bennett et al., 2009)

- Identification and assessment of relationships among multiple ecosystems services by integrated social-ecological approaches.
- Identification of leverage points of small management investments that yield substantial benefits through understanding the mechanisms behind simultaneous response of multiple services to a driver and behind interactions among ecosystem services.
- Strengthening of ecosystem resilience, enhancement of the provision of multiple services, and avoiding catastrophic shifts in ecosystem service provision by managing relationships among ecosystem services.

These propositions aim to build a deeper understanding of how different services are bundled together by key interactions in order to take advantage of synergies among services and avoid unnecessary ecological trade-offs (Bennett et al., 2009). Hence, small changes in the relationships among services can create important opportunities for the management of multiple services. The role of regulating ecosystem services for stability and resilience of the flow of other ecosystem services deserves special attention. Regulating services and ecosystem service interactions involving those have generally been poorly appreciated. Despite the apparent importance of regulating services, environmental management and monitoring have focused on provisioning or cultural services (Carpenter et al., 2006). However, increasing soil biodiversity to increase nutrient availability (Altieri, 1999), conservation tillage improving erosion control (Pimentel et al., 1995), and forest patches enhancing pollination and pest control (Ricketts et al., 2008) are examples where enhancement of regulating services has resulted in improvements in provisioning services. In contrast, enhancement of provisioning services seldom automatically increase regulating services, but instead leads to declines in many cases, e.g. many agricultural techniques for improving agricultural yield have led to declines in other, often regulating, ecosystem services (MEA, 2005).

When natural ecosystems are modified by human activities the focus is primarily on the enhancement of on-site provisioning services in form of agroecosystems. This often comes at the cost of losing regulating services including recycling of nutrients, local micro-climate control, regulation of local hydrological processes, regulation of the abundance of undesirable organisms, and detoxification of noxious chemicals. Biological simplification in agroecosystems can imply quite significant economic and environmental costs, e.g. the need to supply crops with costly external inputs (fertilizer, pesticides etc.) for basic regulating functional components of soil fertility and pest regulation (Altieri, 1999).

In some cases of ecosystem service trade-offs or synergies, management action is not only limited to merely responding to them, but their extend and existence can be actively managed and addressed. Through the establishment of riparian buffers, for instance, the trade-off between agricultural production and water quality by limiting the effect of the driver (fertilizer use) on water quality can be altered without affecting the impact of the driver on agricultural yields (Bennett et al., 2009). The basis for mitigating trade-offs and enhancing synergies among ecosystem services is a sound understanding of the ecological processes that structure the connections between ecosystem services. The potential to create synergies or avoid trade-offs by managing ecosystem service drivers and their interrelationship was revealed by an analysis of Bennett et al. (2009) and is illustrated in Table 4.14. One of their main findings was that agricultural landscapes, in particular, provide many examples of management actions that can either enhance or degrade multiple ecosystem services. Therefore, an integrated view on on-site and off-site ecosystem services in environmental management is promising. However, even in well-studied agroecosystems, knowledge to generalize on when to expect synergies among ecosystem services, how to create them, and how to take advantage of them is lacking (Bennett et al., 2009).

Driver	Service A	Service B	Shared driver	Response type	Interaction type	Synergy or trade-off	Reference
Fertilizer use	Crop production	Water quality	Yes	Opposite	None	Trade-off	Carpenter et al. (1998)
Restoring riparian vegetation	Flood control	Crop production	No	-	Unidirectional, positive	Synergy	Kramer et al. (1997)
Wetland restoration	Flood control	Water quality	Yes	Similar	Unidirectional, positive	Synergy	Zedler (2003)
Afforestation	Carbon sequestration	Water quantity	Yes	Opposite	Unidirectional, negative	Trade-off	Engel et al. (2005)
Dry spells	C sequestr., soil organic matter	Crop yield	No	-	Bidirectional, positive	Bidirectional, positive	Enfors and Gordon (2008)
Cloud forest land clearing	Moisture retention	Carbon sequestr. & tree growth	Yes	Similar	Bidirectional, positive	Bidirectional, positive	Del-Val et al. (2006)

Table 4.14: Examples of ecosystem service relationships and the influence of different drivers; based on Bennett et al. (2009)

Recognizing the different spatial and temporal scales of ecosystem services as well as interrelationships between them and their drivers offers many additional management options and provides incentives for cross-sectoral multi-stakeholder cooperation. Contrary to dealing with on-site and off-site ecosystem services separately, the ecosystem service concept focuses directly on the linkage of both, based on functioning ecosystems. For instance, traditional approaches of on-site regulations of input use (e.g. fertilizers or pesticides) and management practices in agricultural land use are often applied without relation to off-site actions for water treatment. Even when instruments are applied with respect to this functional connection of agricultural effects on water resources, these are often implemented separately for each sector, missing the opportunity to engage stakeholder in cooperation and context-specific solutions.

4.5 Summary

The concept of ecosystem services represents a useful approach for environmental governance by explicitly linking ecosystems' contributions to a variety of human well-being aspects. By focusing on ecosystems as the ultimate source of benefits to all human activities, the concept is inherently cross-sectoral in integrating the human system into the natural system. On the one hand, this facilitates ecosystem-based solutions to environmental management problems in the first place, while, on the other hand, human activities can be assessed according to their impact on the provision of ecosystem services. Therefore, the concept makes a strong case at least for ecological sustainability arguing for more investment in natural capital and specifically the restoration of degraded ecosystems. Understanding ecosystems as the original provider of hydrological ecosystem services can make it attractive to consider them as green infrastructures more systematically (which may be quantified and valued) as an alternative to man-made grey infrastructures. Especially when multiple goals have to be mutually achieved, e.g. flood protection, biodiversity protection and water filtration, these green solutions may reveal advantages.

Moreover, as the ongoing classification of ecosystem services and the understanding of underlying processes highlights, the ecosystem service approach needs sound knowledge of ecosystems, their functioning and the way they provide essential services to any group of society. An important contribution of the ecosystem approach can be seen in its ability to communicate human-natural system linkages, often referred to as an eye-opener (Gordon and Folke, 2000). The MEA, for instance, in principal, had a strong impact on the perception of the interaction between humans and nature. Furthermore, it provides an important link between the challenges in meeting the Millennium Development Goals and the imperative of maintaining biodiversity. Hence, a major objective of the ecosystem approach is to change the way how biodiversity, ecosystems and their services are viewed and valued by society (TEEB, 2010a). Since ecosystem services often represent unaccounted externalities, incentives to maintain ecosystems for continued service provision are low. Additionally, the public or open-access good character of ecosystem services makes it difficult to regulate use levels.

The inherent logic of service flows between provisioning areas and beneficiaries introduces new spatial and temporal interrelationships based on ecosystem properties and human use. Here hydrological ecosystem services provide a plausible and appealing basis to consider hydrological boundaries as basis for the integrated management of water resources. Moreover, hydrological ecosystem services connect different actors within a river basin across sectors and administrative boundaries, based on the idea of provision and benefits of ecosystem services. The role of humans in altering the provision of hydrological services becomes readily apparent and may be considered as a facilitating or disturbing one. Thus, there is the possibility for humans to contribute to positive externalities as well. In the context of IWRM, the provider-beneficiary perspective holds another advantage because it considers the cross-sectoral relationship between land and water uses directly. This has been one of the most difficult but also most essential integration tasks of the IWRM process which has proven especially hard to achieve. Moreover, this perspective allows finding an appropriate management scale based on context-specific characteristics of the natural and human use system by making explicit the linkages between different stakeholders, in particular the users of the resource base (on which the provision of ecosystem services depends) and the beneficiaries of the ecosystem services.

Finally, the ecosystem service concept can illustrate trade-offs and social preferences from many different perspectives such as from on-site and off-site perspectives. Social preferences revealed by valuation processes can be important for the definition and acceptance of environmental goals and an instrument to gain broader societal support for actions implying trade-offs. The communication of non-use values may foster intrinsic motivations for environmental friendly resource use.

Figure 4.14 illustrates how institutional fit and interplay are addressed in an ecosystem service provider and beneficiary context and where trade-offs, i.e. a distribution of benefits takes place.

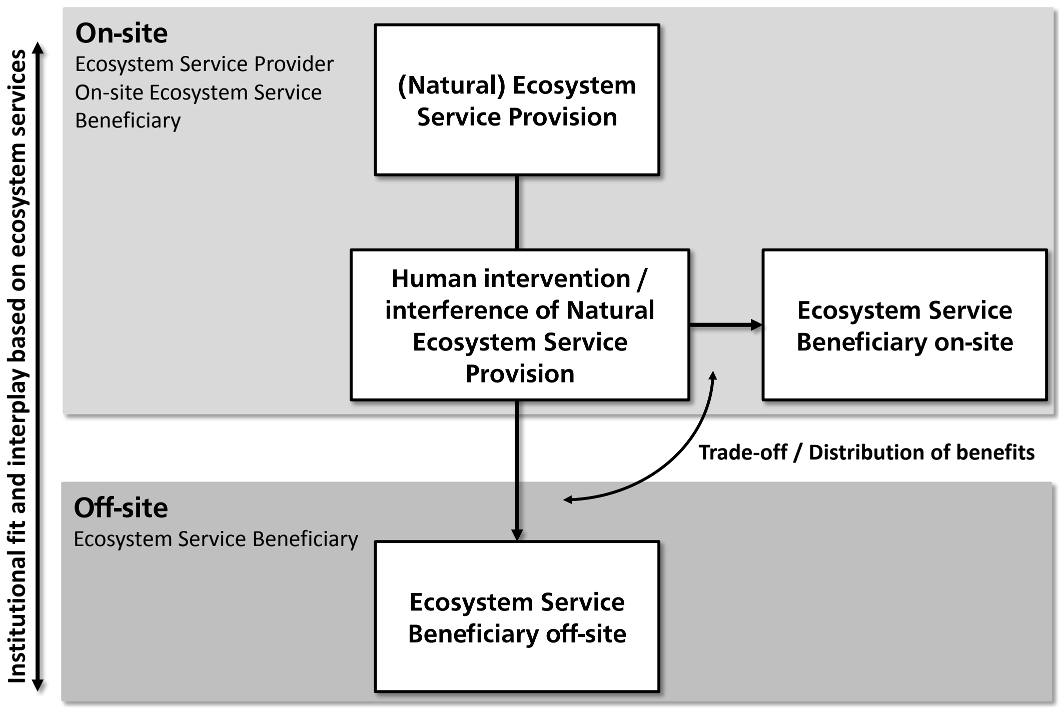

Figure 4.14: Fit and interplay in an ecosystem service provider and beneficiary context (Author's work)

Muradian and Rival (2012) argue, from a policy perspective, that the two critical goals that constitute the core of the governance agenda associated with the ecosystem services approach are (1) to help solve the tension between economic development and environmental conservation and (2) to influence the decisions made by the users of a resource base, so that they align their practices with the interests of the beneficiaries of ecosystem services. These two goals correspond to two areas of action, one of "creating linkages between different [institutional] layers and stakeholders in order to deal with complex economic, social and ecological inter-dependencies, and [two] of inducing changes in the use or the property rights of the resource base that provides the concerned services, so as to align the interests of different social agents" (Muradian and Rival, 2012). Meanwhile the ecosystem service concept is increasingly applied as a policy instrument for the governance of environmental, especially water, resources in developed and developing countries.

The Council on Environmental Quality (CEQ) of the Executive Office of the President of the United States of America, for instance, acknowledges in its recently published *Principles and Requirements for Federal Investments in Water Resources* the importance of ecosystem functions and resulting services. Their economic importance is highlighted with the following statement: "Healthy and resilient ecosystems not only enhance the essential services and processes performed by the natural environment, but also contribute to the economic vitality of the Nation“ (CEQ, 2013). Besides stressing the economic importance of healthy ecosystems for the public, the CEQ emphasizes the utility of ecosystem functions as green infrastructures; especially the role of floodplains is highlighted. Green infrastructure solutions are assigned priority over man-made infrastructures in achieving the objectives of water resources management. Moreover, the *Principles and Requirements for Federal Investments in Water Resources* stresses that the scope and scale of water resources management should reflect the nature of cause and effect relationships between effects on ecosystems and resulting public benefits (CEQ, 2013). Thus, the identification of an appropriate unit for the management of water resources is proposed to be based on hydrological ecosystem services. This policy change towards a stronger focus on ecosystem functions and services is accompanied by a call for more adaptive management. This is not surprising as the ecosystem service concept incorporates socio-ecological learning as advocated by adaptive management.

The application of the ecosystem service concept in water resources management is also on the rise in the EU where it has been identified as one of the pillars of the assessment of impacts in the preparation of the 2012 Commission's Blueprint to safeguard the future of European Waters by 2015 (Maes et al., 2012). The EU recognizes as well the importance of investing in nature as a source of economic development in its regional and cohesion policy (EC, 2011b). Furthermore, a new proposal for the EU's Common Agriculture Policy identifies restoring and preserving ecosystem services as one of six priorities (EC, 2011a).

In the context of developing countries, the International Union for Conservation of Nature "Water and Nature Vision" identified an *ecosystem approach* as a key element and new paradigm when implementing IWRM within river basins (Smith and Cartin, 2011). This *ecosystem approach* is defined as a strategy for integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way. While meeting people's needs as a central element of the ecosystem approach it aims to:

- Maintain ecosystem functions and services
- Enhance equitable sharing of benefits
- Promote adaptive management strategies
- Implement management actions through decentralization
- Foster intersectoral/interdisciplinary cooperation (Smith and Cartin, 2011)

The ecosystem service concept is increasingly promoted through the policy instrument of Payments for Ecosystem Services (PES), i.e. Payments for Hydrological Ecosystem Services (PHES). In the following chapter, this policy instrument is presented and its role as a complementary policy instrument to improve spatial fit and institutional interplay in the context of IWRM operationalization is analyzed.

5 The concept of Payments for Hydrological Ecosystem Services

After the rise of the ecosystem service concept in the 1970s and 1980s (Schumacher, 1973; Westman, 1977; Ehrlich and Ehrlich, 1981) and its mainstreaming in both science and policy discourse in the 1990s through several influencing publications (Daily, 1997; Costanza et al., 1997), the concept found its way in the late 1990s and 2000s into a broader policy arena and first practical application. Prevailing examples of the increasing policy relevance of the ecosystem service concept are the adoption of the *Ecosystem Approach* in the Convention on Biological Diversity (UNEP, 2000), the MEA (2005), the Stern Review on the Economics of Climate Change (Stern, 2006), and the Economics of Ecosystems and Biodiversity initiative (EC, 2008). With increasing research and public communication on economic values of ecosystems and their services to society, interest has grown in the design of Market Based Instruments (MBI) to create economic incentives for conservation and improve natural resource management based on the ecosystem service concept (Daily and Matson, 2008; Jack et al., 2008). Markets for Ecosystem Services (MES) (Bayon, 2004) and Payment for Ecosystem Services (PES) schemes (Landell-Mills and Porras, 2002; Wunder and Albán, 2008) have been developed as specific tools to provide these incentives. Examples for Markets for Ecosystem Services are emission trading schemes of greenhouse gases in the EU, United Kingdom, and the Chicago Climate Exchange in the USA, as well as trading schemes for sulfur dioxide emissions through the Clean Air Act and Wetland mitigation banking in the USA (Stavins, 2000; Robertson, 2004; Bayon, 2004). MES are often considered as a special type of PES characterized by their free-wheeling transactions within an established marketplace of potential ecosystem service sellers and potential buyers (cf., Gutman, 2003).

According to Pesche et al. (2013), the concept of Payments for Ecosystem Services[1] originally arose from changes in perceptions of the efficacy of traditional nature conservation policies in developing countries with high levels of biodiversity. Two aspects had special influence on the speed of PES evolution. On the one hand, the environmental policies applied during the late 1980s and 1990s, based on the rationale of conservation through development (mostly pursued through Integrated Conservation and Development Projects; ICDP), were questioned in their efficacy and at the end of the 1990s better targeting and direct payments, i.e. direct investments, for achieving the desired policy outcomes were proposed instead (Simpson and Sedjo, 1996; Ferraro and Kiss, 2002; Simpson, 2004). Rice et al. (2001), for instance, argue that direct payments are more effective than, for instance, trying to promote sustainable, selective logging, which has been a feature of many integrated conservation and development projects. Moreover, the absence of clear links between performance and benefits has been identified as a specific weakness of ICDP (Wunder, 2005). On the other hand, the lack of sustainable funding and the need for additional sources of funding led to the search for innovative funding mechanisms (Pesche et al., 2013). Finally, a first systematic connection between ecosystem services and markets for their provision was made by Landell-Mills and Porras (2002).

Hence, since the end of the 1990s Payments for Ecosystem Service schemes have been established for various hydrological services related to the quality, quantity, or timing of freshwater flows from upstream areas to downstream users and carbon sequestration mainly in Central and South America, for habitat conservation for wildlife in Africa and South America, for bio-prospecting in Costa Rica, and for the provision of incentives for agro-environmental measures especially in the EU and the USA (Bond and Frost, 2006; Corbera et al., 2007; Wunder and Albán, 2008; Asquith et al., 2008; Pagiola, 2008; Dobbs and Pretty, 2008). The country-wide program *Pago por Servicios Ambientales*, initiated in 1997 in Costa Rica, was the first application of a formal PES mechanism in a developing country (Pagiola, 2008). The principal aim of the program is to reverse severe deforestation rates which have been experienced in the past. In the early 2000s, a growing number of PES followed throughout Meso-American and South American countries (Corbera et al., 2007; Kosoy et al., 2007; Asquith et al., 2008; Pagiola, 2008; Wunder and Albán, 2008). Rather recently, cap and trade programs such as Reduced Emission from Deforestation and Degradation (REDD) are being discussed as possible international PES schemes for the post Kyoto protocol (Gómez-Baggethun et al., 2010). However, many early PES schemes focused on the role of forest ecosystems for the provision of several services, most of all carbon sequestration and hydrological services. Thus, these PES schemes often had primarily conservation objectives, i.e. the objective to find additional funding sources for forest protection (cf., Johnson et al., 2001). Especially in Latin America, detaining deforestation was a major concern and a motive for looking for alternative policy instruments since traditional policies based on command and control mechanisms did not succeed (Pagiola et al., 2002).

Contrary to command and control instruments, PES are intended as direct and flexible incentive-based mechanisms where a user (i.e. beneficiary) of an ecosystem service makes a direct payment in cash or in kind to an individual or community whose decisions on the use of natural resources have an impact on the ecosystem service provision (OECD, 2010; Carius, 2012). Through the incentive, it is intended to achieve a behavioral change on decisions of land and resource uses in order to reduce ecosystem service loss or enhance their provision. Thereby, PES schemes aim to address

[1] Payments for Environmental Services or the terms rewards, compensations, stewardship or reciprocal agreements instead of the term payments are also commonly used. Hereafter the terms payment and compensations will be used as synonyms.

market failures by translating external non-market benefits (external effects or externalities)[2] of ecosystems into tangible incentives for their provision (OECD, 2010). Thus, besides individual on-site benefits of resource and land use, off-site benefits to a potentially larger group of beneficiaries may as well be more explicitly considered in decision-making through an internalization of externalities. Moreover, the conditions in terms of payment amount and restrictions of utilization of land providing the ecosystem service can, in theory, be adjusted individually between service provider (seller) and service user (buyer) providing a more flexible and efficient approach to achieve environmental management goals.

The principal logic of PES is illustrated in Figure 5.1, based on an example of different use options of agricultural land and respective benefits to the land user and external costs to off-site water users. When compared to conservation agriculture, the land user receives greater (net) private benefits through conventional agriculture. This conventional agricultural land use, however, incurs costs to downstream water users or beneficiaries in the form of reduced watershed services.

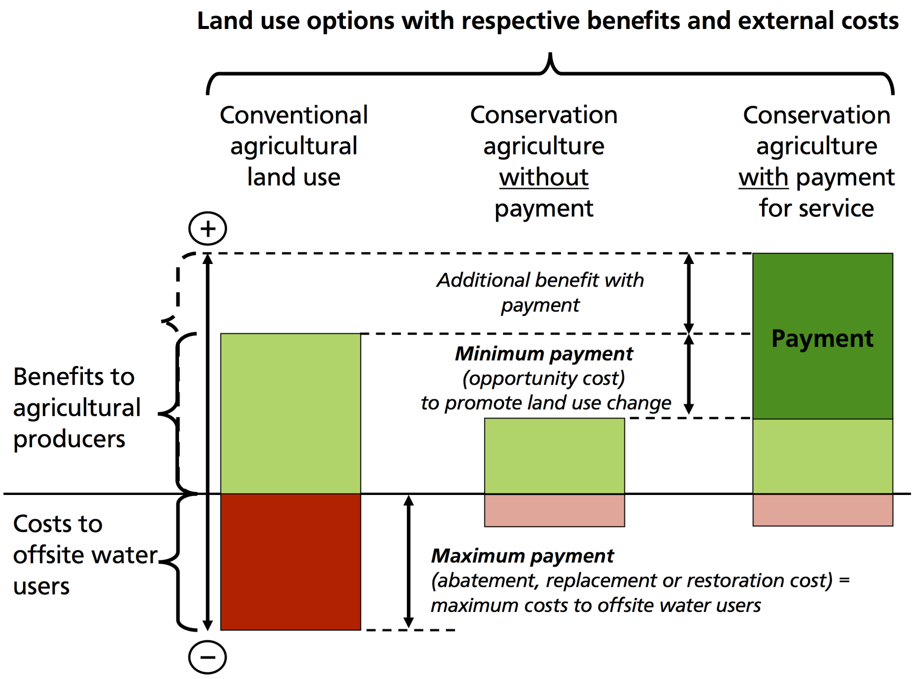

Figure 5.1: Logic of Payments for Hydrological Ecosystem Services schemes; from Hack (2010) based on Pagiola and Platais (2002)

From a straight economical point of view, the *minimum payment* that the land user will be willing to accept as compensation to change his land use focusing more on conservative aspects favoring downstream service provision is the foregone opportunity cost of the alternative land use. At the same time, the *maximum payment* that an ecosystem service beneficiary is willing to pay for conservation is the total cost of damage incurred when the land is used in the conventional manner. Hence, the maximum payment principally considers restoration costs, but may as well consider replacement or avoided costs, e.g. the cost of improving water quality through engineering structures. If, as in the case illustrated schematically in Figure 5.1, the potential benefits of conservation are larger than the minimum payment, a PES mechanism can potential be mutually beneficial. Since participation in PES is voluntary for the land user, the economic theory suggests that rational land users will enter into an agreement as long as the payments cover at least their opportunity costs of changing their land use practices (Kosoy et al., 2007). Accordingly, any payment level between the minimum and the maximum is supposed to provide enough incentive to induce a change in land use towards greater ecosystem service provision. Ideally, through such an agreement both the individual land holder (in a micro-economic sense) and the broader society derive (macro-economic) benefit from the provision of ecosystem services from the contracted lands. This would represent a pareto-optimal solution. Thus, the fundamental problem of conservation that the costs of abstaining from degrading types of utilization, i.e. non utilization, are incurred by the user of the land, while others would benefit from conserving the resource is attempted to be solved through an equivalent compensation. The costs of abstaining from destructive forms of utilization include the costs of safeguarding non-degrading utilization, and the opportunity costs, for instance foregone income from intensive agricultural utilization. As the conservation associated with the provision

[2] Such market failures caused by externalities are largely acknowledged as an explanation for the decline of important ecosystem services as a result of human pressures (cf. Westman, 1977; Hanley and Spash, 1993; Loomis et al., 2000; MEA, 2005; TEEB, 2010a).

of ecosystem services of his land implies a trade-off in terms of reduced individual gains, the land user has less interest in conserving the land than the beneficiaries of the ecosystem service provided. Hence, the incentive is provided by the beneficiaries (or an intermediary) to motivate and compensate the right holder to provide the service. In an ideal case, the level of compensation corresponds to both the benefit that those beneficiaries making the compensation derive from the non-occurrence of resource degradation and at the same time to the net costs of non-utilization incurred by the right holder of the land (monitoring and opportunity costs of abstaining from degrading local utilization minus the own benefit of the right holder from resource conservation).

Thus, taking a market perspective assumes that a payment between the minimum and maximum payment has distributional and cost-effective implications, but will bring about the same environmental change (OECD, 2010). Figure 5.2 illustrates this by looking at marginal costs and benefits. The theoretical optimal provision of ecosystem services is given at Q^* in the figure, where the marginal costs of service provision and the marginal social benefits are equal. The costs to provide the service encompass the opportunity costs of the alternative land use incurred by the land user and the transaction costs associated with the implementation of the PES mechanism. For this optimal case, the area defined by points *yxb* in the figure illustrates the beneficiary's surplus (i.e. consumer surplus in market economics) and the area *ybk* represents the provider's surplus (i.e. producer surplus). However, the actual level of ecosystem service provision without payment is illustrated with point Q^1 as a consequence of the presence of market failures and the resulting divergence between private and social marginal benefits. Hence, a payment P^* is required to internalize the beneficiaries costs, i.e. to correct for the market failure, and achieve a more socially optimal level of ecosystem service provision. According to Engel et al. (2008), in practice the sum of offered payments may be insufficient to attain Q^* either because of the presence of incentives for beneficiaries to free-ride (obtaining the benefits without paying for them), or because financial resources (e.g. from government) available for ecosystem service conservation and sustainable resource use are limited. In this context PES aim at an improvement above the status quo of socially sub-optimal service provision. In order to achieve a level of service provision where the marginal social benefits are greater than the marginal costs of provision (illustrated with point Q^2), the payment (i.e. price) can be set between the minimum payment (at P^{MC}) and the maximum payment (at P^{MB}) as discussed above and illustrated in Figure 5.1. Point P^{MC} indicates a payment amount according to the marginal cost of provision and allocates the greatest welfare surplus to the beneficiaries (area defined by points *wmxz*), while area *wzk* represents the private land user's surplus. In contrast, a payment at point P^{MB}, equal to the marginal social benefits, represents an allocation of greater welfare surplus (area *wmnk*) to the land user and less surplus to the buyer (area *mxn*). When aiming to maximize the cost-effective achievement of social benefits, the minimum payment is equal to the land user's marginal costs of service provision. Assuming that Q^1 is provided by existing incentives for the land user (representing the baseline level of service provision), a payment is only required to purchase additional ecosystem service benefits (moving from Q^1 to Q^2). The most cost-effective payment at P^{MC} requires finance for the difference from areas *vwzt* and *rdot*. In turn, payments have to sum up to areas *vwzt* and *wmnz* minus *rdot* in the case of a payment equal to the marginal social benefits, P^{MB}, the maximum payment. Therefore, when favoring social benefits, cost-effectiveness increases as the price moves towards point P^{MC}. Depending on the magnitude of the ecosystem service benefits provided and the different (opportunity and transaction) costs incurred in their provision, the levels of P^{MC} and P^{MB} are likely to vary from one site to another as a result of spatial heterogeneity (OECD, 2010).

Theoretically, the conditions to join a PES are met if the payment is marginally lower than the level of the benefits derived by the potential payers through conservation of the resource or marginally lower than the level of damage which the potential payers would suffer as a result of degradation of the resource.

PES schemes, as opposed to the broadly accepted polluter pays principle, follow a provider gets or beneficiary pays logic. Although being different in their underlying logic, both principles can be applied together without contradiction. By defining a reference level of an environmental standard, for instance, the polluter pays principle can be applied in case the environmental performance is below this level and for any measure of improvement above the required reference level the provider gets principle may be applied. This, for example, is done in the European agricultural policy. Although the intention of the polluter pays principle is to charge polluters for the costs of pollution abatement, in practice this is often not implemented properly. The fees for waste water discharges based on the polluter pays principle, for instance, generally do not relate to waste water and drinking water treatment costs, which are often subsidized (Stallworth, 2003). In developing countries, however, the polluter pays principle is often difficult to apply because the polluter may be poor and therefore unable to pay. In case of agricultural activities, often practiced for subsistence, the pollution from these activities is, on the one hand, very difficult to assess because of their diffuse nature and, on the other hand, subsistence farmers do not have an alternative.

Moreover, there is an important implication on the designation of roles, when applying the polluter pays or provider gets principle. Although a direct relationship between the polluter and the polluted can be established in both situations, i.e. between the provider and the beneficiary, in practice, however, a polluter does not pay the cost of pollution to the polluted but to the government according to regulatory sanctions. Cordato (2001) argues therefore that “in such cases, the polluter pays principle is used to promote an environmental agenda rather than to insure that real polluters pay

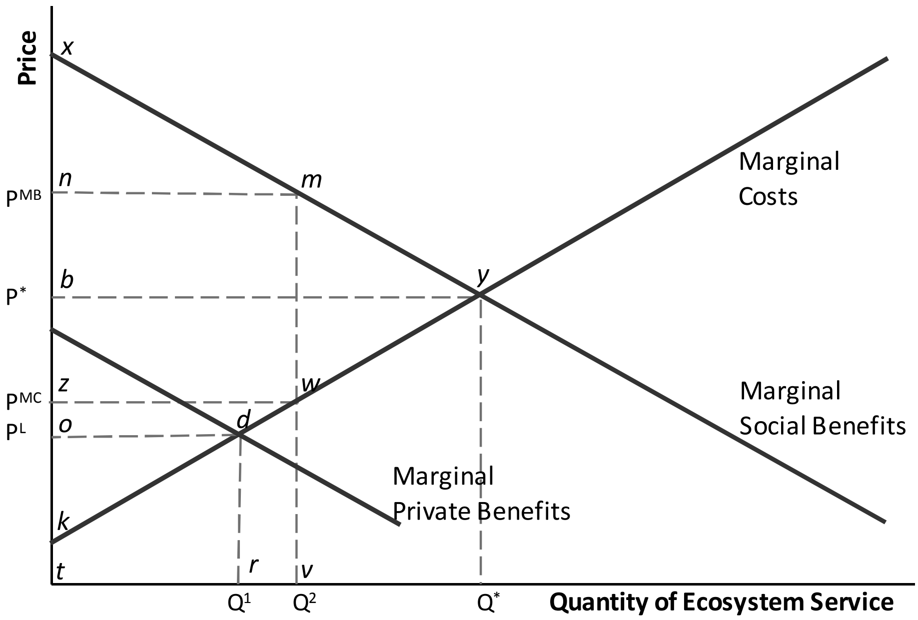

Figure 5.2: Optimal provision of ecosystem services, and the distributional and cost-effectiveness implications (OECD, 2010)

compensation to real victims of their activities". In contrast, the payments of beneficiaries to providers are often, although not always, made in a more direct manner. This difference is due to the voluntary nature of the provider gets principle as opposed to the mandatory polluter pays principle. The provider gets principle has a further advantage in its psychological function of encouraging doing what is expected to be environmentally sound and therefore morally accepted by society (cf., Fehr and Falk, 2002). This is especially important when communicating the importance of non-use values or use values of others, directly impacted by externalities. An example for an economic instrument following the polluter pays principle are environmental taxes, e.g. charges on environmentally-damaging activities. Thus, while the polluter pays principle provides disincentives, often in form of command and control regulation, which mandates actors to undertake specific actions and applies sanctions if they do not comply, the provider gets principle provides incentives in order to stimulate environment-friendly behavior based on voluntary agreements. However, the use of positive incentives does not imply the complete absence of negative incentives within a PES intervention. Sommerville et al. (2009) argue if payments are used to influence attitudes and participation in a regulatory environment, the repercussions associated with regulation can act as a negative incentive. Moreover, PES schemes may also imply negative incentives through coercion to participate based on social pressure from other community members, or even through regulatory fines and punishments (Sommerville et al., 2009). Notwithstanding, for those participating in a PES schemes positive incentives have to outweigh the negatives. Participation and engagement of ecosystem service providers is based on the transfer of positive incentives to these individuals. However, positive incentives go beyond monetary payments and payments in-kind as social cooperation, local norms, or religious beliefs can also influence behavior positively (cf., Deci and Ryan, 1985; Henrich et al., 2001). Hence, Sommerville et al. (2009) point out that, in addition to monetary transfers, incentives may as well come from social impacts such as tenure legitimacy and pride. In PES schemes where individuals choose to provide services at payments lower than their opportunity costs (Wunder, 2005; Kosoy et al., 2007), the interaction of different incentive forms becomes apparent. In a silvopastoral PES scheme in Nicaragua, Van Hecken et al. (2012) observed that while farmers welcomed the payments, they attributed more importance to technical assistance and a growing market demand for their products as the basis for their decision to alter their land use. Garbach et al. (2012) too, highlight the importance of technical assistance to improve agricultural practices as having a positive influence on participation in other PES schemes. Accordingly, Van Hecken et al. (2012) argue that identifying a socially and culturally acceptable type of payment (monetary or in-kind) that is considered to be fair by the local community should be an essential part of any PES negotiation process. Studies of Sommerville et al. (2010) and Gross-Camp et al. (2012) suggest that a perceived fairness of the distribution of the costs and benefits among participants is a key determinant of local acceptability of PES schemes (cf., Adhikari and Boag, 2013).

However, in some situations there may be no other option than the provision of positive incentives. This is the case, for instance, when land users possess legal control over service provision, e.g. they have the right to carry out certain land-use changes that would change service provision. Here incentives may influence the decision to produce the service, while disincentives through restrictions are not an option because there is no legal justification. A typical situation would be when land users are paid to implement certain farming practices, e.g. to develop or maintain hedgerows (Dobbs and Pretty, 2008). Most user-financed Payments for Hydrological Ecosystem Services (PHES) schemes aiming to safeguard water quality through particular land-management practices were developed because positive incentives were the only feasible option to influence service provision (Sommerville et al., 2009). It is also argued that positive incentives, as provided through PHES schemes, are used to influence attitudes toward a regulation or a change in legal enforcement. According to Pagiola (2008), a principal motivation for the establishment of Costa Rica's PES program was to gain acceptance for a legislative ban on deforestation on private lands from landowners and to attract their cooperation. When using positive incentives, however, the scale and the actors addressed matter. In order to influence the decisions individuals make on how they use resources, these individuals need to be addressed directly by the incentives rather than making payments to a regional government. This aspect, among others, will be further discussed in the following section in the context of directness.

5.1 Economic conceptualizations of Payments for Ecosystem Services

The following discussion about the economic conceptualization of PES is quite important as it defines the aim and scope of the instrument as well as the distribution of roles between different actors, i.e. the degree of involvement of public and non-public actors within a policy mix.

Most of the literature referring to PES as a market based or market-like mechanism follows Wunder (2005) in defining PES as

(a) "a voluntary transaction where
(b) a well-defined environmental service (or a land use likely to secure that service)
(c) is being *bought* by a (minimum one) service buyer
(d) from a (minimum one) service provider
(e) if and only if the service provider secures service provision (conditionality)".

The theoretical background of this popular PES definition is an attempt to put the *Coase Theorem*[3] into practice by integrating ecosystem services into markets, dealing with ecosystem services as with any other market transaction (cf., Engel et al., 2008). The rationale behind the Coase Theorem is that if private property rights are clearly defined by enforceable contracts, then the generator and recipient of an externality can, through voluntary exchange, potentially reach an agreement that maximizes social welfare. Hence, besides the enforcement of property rights, government intervention is not required to achieve optimum social welfare. Thus, effective environmental governance is achieved by creating markets for trading ecosystem services that lead to *getting the price right* that formerly caused market failures and under-supply of ecosystem services. However, in order to achieve this, wealth effects and transaction costs need to be absent or insignificant (cf., Coase, 1960).

Accordingly, proponents of this Coasean approach to PES generally stress the definition of property rights, creating enforceable contracts, and reducing transactions costs, though many still see the requirement for some degree of government intervention (Farley and Costanza, 2010). According to Engel et al. (2008), Coasean PES approaches are "likely to be efficient, as the actors with the most information about the value of the service are directly involved, have a clear incentive to ensure that the mechanism is functioning well, can observe directly whether the service is being delivered, and have the ability to re-negotiate (or terminate) the agreement if needed". Thus, proponents of economic conceptualizations of PES in this sense highlight the generally more positive effects on economic efficiency and environmental effectiveness as compared to Pigouvian[4] approaches such as environmental taxes to economically internalize externalities. Since fiscal efficiency is a main focus of this approach, many proponents frequently claim private sector PES schemes to be more promising than public ones (Wunder, 2008).

However, in practice, PES usually include a mix of market elements, e.g. voluntary participation, individual cost-benefit considerations, and hierarchical elements, such as legal property rights, institutional rules and contracts. The diversity of PES schemes in terms of different scales (e.g. local/regional, national or global), funding sources (e.g. private or governmental payments) or other characteristics has been stressed by several authors of global PES reviews (Johnson et al., 2001; Landell-Mills and Porras, 2002; Gutman, 2003).

The divergence from the theoretical foundations of the Coasean Theorem in the practical establishment of PES has been addressed, among others, by Jack et al. (2008); Muradian et al. (2010); Vatn (2010); Muradian and Rival (2012); Schomers and Matzdorf (2013). According to their findings, there are hardly any PES schemes that represent a purely voluntary market transaction without governmental involvement. Vatn (2010) describes several aspects of PES schemes which imply either a more Coasean or a Pigouvian-like concept. One distinguishing aspect is rights and rules to land

[3] Ronald Coase, a British economist, introduced the concept of transaction costs to explain the nature and limits of firms (Coase, 1937) and suggests with the Coase Theorem that with well-defined property rights the problems of externalities could be overcome (Coase, 1960).

[4] The economist Arthur Pigou argued that a tax should be applied to market activities that are generating negative externalities in order to correct an inefficient market outcome (cf. Pigou, 1920).

property and use. In order to make a payment the provider has to be known and holding the rights to change the use of the land. These rights can actually be formal or informal, although the latter will increase the risk of actual provision for the buyer. From the current use of the land and the desired change in land uses the question arises whether its actual use conflicts with existing rules (e.g. formal regulations or informal use rules) and whether the polluter-pays-principles (in case of illegal land use) or the provider-gets-principles (in case of additional services provided) should be applied. Vatn (2010) argues that various mixed responsibilities are possible between these extremes. A specific practice or environmental quality as the reference point, sound forest practices for instance, could be agreed on while delivering a higher level of services would result in being paid and delivering less would imply having to pay oneself (Vatn, 2010); similar to the duty of care concept of positive and negative price signals described by Young (2000). Vatn (2010) reasons further that "according to the literature describing various MES / PES systems, providers of environmental services seem to be implicitly and exclusively granted the right to the status quo uses, i.e. a form of provider gets with rarely no discussion about whether these uses are legitimate."

Schomers and Matzdorf (2013), like Vatn (2010), highlight differences in the economic conceptualization of PES and distinguish between Coasean and Pigouvian PES schemes as well as financial incentives beyond Coase and Pigou. Although pure Coasean PES examples are hardly found in the literature, Schomers and Matzdorf (2013) refer to them as "cases where benefits from ecosystem management are provided at local scales". Typical examples of Coasean PES are mechanisms where "downstream water users pay upstream land stewards for land use changes that are assumed to increase both, water quality and quantity" (Schomers and Matzdorf, 2013). However, since local municipalities are often involved to varying degrees from setting up to running the scheme as intermediaries, service buyers or as well service providers, not all of them fully comply with the Coasean conceptualization. Moreover, repeatedly payments from beneficiaries are not realized on a voluntary basis, e.g. in the case of water fees, and in some cases contracts are not negotiated privately among relevant stakeholders. These issues conflict with the Coase Theorem and the PES definition by Wunder (2005). Table 5.1 summarizes Payments for Hydrological Ecosystem Services (PHES) schemes categorized by Schomers and Matzdorf (2013) as either more Coasean or more Pigouvian.

On the other hand, several authors have referred to PES schemes led by (national) governments as a Pigouvian economic conceptualization (Vatn, 2010; Pattanayak et al., 2010; Van Hecken and Bastiaensen, 2010). Although Schomers and Matzdorf (2013) argue that governmental payment approaches rather follow the environmental pricing and standards procedure (cf., Baumol and Oates, 1971), they use the Pigouvian economic conceptualization as a category to distinguish them from more Coasean conceptualizations. Van Hecken and Bastiaensen (2010) describe the Pigouvian conceptualization as "taxing negative or subsidizing positive externalities within existing product markets".

Just as the Coase Theorem, the Pigouvian concept attempts to correct an apparent market failure by the internalization of externalities. Hence, in case of negative externalities of a market activity a specific tax is applied equal to the negative externality in order to cover the social costs caused by the market activity. Often environmental pollution is addressed with Pigouvian taxes. In the context of PES, the focus is on positive externalities provided through a certain land use following a provider gets logic. Given this proposition, a Pigouvian subsidy is provided by the state to pay the providers of the positive externality. While the Pigouvian subsidy requires that the payment equals the marginal net benefit that it is supposed to generate, the environmental pricing and standards procedure, in contrast, "begins with a predetermined set of standards for environmental quality and then imposes unit taxes (or subsidies) sufficient to achieve these standards" (Baumol and Oates, 1971). Therefore, a defined payment is established reflecting the price for the provision of public goods. As payments in governmental PES are not necessarily linked to a (already marketed) commodity which is assumed to provide the beneficial externality, Van Hecken and Bastiaensen (2010) recognize that these schemes differ in this sense from Pigouvian subsidies. Instead, the externality (i.e. ecosystem service) itself is traded as a commodity (cf. Kosoy and Corbera, 2010; Van Hecken and Bastiaensen, 2010) with the state as "third party acting on behalf of service buyers" (Engel et al., 2008). This illustrates the principal difference between Coasean and Pigouvian conceptualizations as being the directness of transaction (Schomers and Matzdorf, 2013). According to this differentiation, only the Coasean conceptualization establishes a direct interaction between the buyer and the service provider through the agreement on a transaction related to a behavioral change. Vatn (2010) ascribes this differentiation of the two types of conceptualization to different exclusion cost structures, i.e. the public good characteristics of the service. The beneficiaries of Coasean PES are mainly located at a local or regional proximity to the provisioning area and can therefore be easier and more directly identified, for instance in an upstream-downstream relationship. In the case of Pigouvian PES this is often different and the state pays for the provision of services to the general public. Table 5.1 provides examples of Pigouvian PES discussed by Schomers and Matzdorf (2013).

In the context of the discussion on Coasean and Pigouvian PES it is important to consider the underlying discourse on governmental versus market-like institutions. In reality, as has been recurrently stated, purely market-based solutions of PES are absent. Instead it has to be recognized that "PES are commonly imbedded in a broad institutional setting with an actor constellation that does not resemble simple market-based buyer and seller relations" (Schomers and Matzdorf, 2013). Thus, considering either Coasean or Pigouvian does not reflect the broad diversity of institutional settings of existing

Coasean conceptualization		
Examples	**Description**	**References**
Paso de Caballos River Basin, Nicaragua	Upstream landowners are paid by downstream households for reforestation and conservation efforts. Households created a Water Committee and negotiate contracts with upstream land users	Corbera et al. (2007)
Escobas River Basin, Guatemala	Upstream forest conservation is financed by a local hydroelectricity and water company that benefits from continuous water flows and reduced sediment loads. Payments are financed by an increased water tariff	Corbera et al. (2007)
Cidanau River Basin, Indonesia	State-owned water company signed contracts for watershed conservation with upstream farmers	Leimona et al. (2010)
Palaurco River Basin, Ecuador	Local municipality charges obligatory water fee to downstream water-using households. The fee is paid via a water fund to upstream landowners, who are contractually committed to halting deforestation and naturally regenerate degraded agricultural lands	Quintero et al. (2009); Wunder and Albán (2008)
Los Negros River Basin, Bolivia	Compensation by local municipality targets on curbing upland deforestation to overcome the growing problem of water scarcity	Asquith et al. (2008)
Pigouvian conceptualization		
Examples	**Description**	**References**
National PES Program, Costa Rica	Private forest landowners are paid for forest conservation or reforestation by the state. Financing comes mainly from a mandatory tax on fossil fuels, donor and domestic water suppliers contribute as well	Pagiola (2008)
Payments for Hydrological Ecosystem Services Programme, Mexico	Federal government uses an obligatory water fee to finance the conservation of existing forests using a uniform payment scheme	Southgate and Wunder (2007)
Conservation Reserve Program, USA	Federal government purchases environmental services from agriculture halt conversion of wetland and highly erodible land to cropland	Dobbs (2006)
Sloping Land Conversion Program, China	Federal government pays for conversion of sloped cropland to grasslands or forests to reduce erosion and alleviate poverty	Gauvin et al. (2010)
Working for Water Program, South Africa	Federal government pays unemployed to clear invasive plants and to restore natural fire regimes in mountain catchments and riparian zones	Swallow et al. (2010)

Table 5.1: Economic conceptualizations of Payment for Hydrological Ecosystem Service schemes; based on Schomers and Matzdorf (2013)

PES schemes. Accordingly, Van Hecken and Bastiaensen (2010) stress that "PES is mistakenly understood as a simple matter of financial incentives" and should "be understood as part of a broader process of local institutional transformation rather than as a market-based alternative for allegedly ineffective government and / or community governance". Hence, authors increasingly highlight that PES should be regarded as a complement rather than a substitution to regulatory instruments (Sommerville et al., 2009; Vatn, 2010; Van Hecken and Bastiaensen, 2010; Muradian et al., 2010), considering the interplay of the whole institutional setting crucial for their success (cf. Greiber, 2009; Corbera et al., 2009; Muradian and Rival, 2012; Schomers and Matzdorf, 2013).

The debate on granting different property rights in form of different use rights and whether to apply the polluter pays or the provider gets principle, for instance, is a typical question of institutional interplay in PES implementation in practice. This debate can be very fruitful, when carried out deliberately, in order to clarify present formal and informal rules and how to deal with them. Here it becomes clear that PES are actually applied within a policy mix and that different policy instruments are needed for different problems. This way institutional interplay is promoted through the communication of existing formal institutions as well as by stakeholder involvement and public participation in clarifying the "rules of the game", i.e. mutually agree on which land and water uses are socially acceptable and which are not. This can build the basis for agreements on what actions deserve to be rewarded through compensations (e.g. payments) and what actions need to be regulated by command and control through bans and sanctions. Likewise, Vatn (2010) addresses the fact that PES schemes are not created in an "institutional vacuum". Because of all the uncertainties and control problems involved in the case of ecosystem services, well-functioning PES schemes demand cooperative parties, hence, taking the wider institutional context seriously is important for success. In this context, social perceptions and values play a fundamental role. Hence, in order to agree on values and payments for certain actions, trust building and participation are vital elements of PES implementation (Corbera et al., 2007).

Carius (2012) comes to a similar conclusion with regard to the substantial diversity of PES differing in their scale, organizational form, service specification, payment and monitoring models as well as participants. Therefore, he stresses that the form of the PES mechanism "depends on regulations and norms as well as relevant political, social, economic and ecological conditions in a specific location". On the one hand, this highlights the dependence of PES on specific prevailing conditions but, on the other hand, also the flexibility of PES mechanisms in adapting to context-specific constellations of human and natural system interactions and specific institutional settings.

Muradian et al. (2010) discuss the Coasean conceptualization of PES in the context of practical implementation constraints and discrepancies from the Coasean theory. They refer in their discussion to "the complex contexts in which most PES schemes operate - particularly in developing countries - in order to draw insights for the elaboration of an alternative conceptual framework" (Muradian et al., 2010). The situations that PES application faces are generally unfavorable for a strictly market-based Coasean approach. The criticism of Muradian et al. (2010) on such an approach stresses that in order to be effective a Coasean approach requires absent or at least very low transaction costs, clearly assigned property rights, sufficient information for market-functioning as well as absence of the state. However, in general, there is high uncertainty in PHES schemes due to scarce information and knowledge on cause-effect relationships between land use and hydrological ecosystem service provision (see Section 4.2). This often results in practitioners and policy-makers not having full access to vital information (Muradian et al., 2010). This *information gap* (see Section 3.3) is an intrinsic problem of IWRM operationalization and instruments addressing this problem explicitly are needed. According to Muradian et al. (2010), in order to facilitate a realistic connection between payments, services and economic benefits, a *genuine* PES requires developing sound, context-specific, socio-ecological research prior to its implementation. This research is carried out to different degrees in practical implementation, thus, implying different transaction costs which may conflict with the Coasean theory. However, for many local PHES schemes this socio-ecological research is realized while building the basis for a later PHES scheme to develop.

Furthermore, the monitoring of actual ecosystem service provision or the accomplishment of other agreed ecological goals is often not well developed and rather builds on the faith of the participants (Muradian et al., 2010). An important aspect in the context of IWRM is that both PHES and IWRM, i.e. water resources management in general, require socio-ecological information in order to increase their ecological and economical effectiveness. PHES schemes, partly because of their inherent ecosystem approach logic, though, provide the incentives to increase this knowledge and find ecosystem-based solutions for economic trade-offs. Moreover, in local PHES schemes information, i.e. what sometimes is considered *faith*, involves local knowledge on ecosystem functioning. This can be very valuable, for instance, in the context of adaptive management (see Section 2.4). Institutional interplay is often promoted in local PHES schemes with the establishment of technical and managing committees. Through the participation of different sector organizations (e.g. for forest management or improved agricultural practices) in technical committees horizontal institutional interplay is facilitated.

Muradian et al. (2010) argue that the trade-off between the need to estimate efficiency gains resulting from the intervention and the need to keep transaction costs low enough to make PHES schemes feasible challenges the Coasean conceptualization and instead highlights the need to make decisions in a context of incomplete information. However, implementing PHES with a broader objective than efficiency alone can still make them feasible.

Another point refers to the social embeddedness and stakeholder perceptions in PES schemes. Clements et al. (2010), based on a comparison of three PES schemes for wildlife protection and ecotourism in Cambodia, recognize the importance of intrinsic motivation to determine behavior and endogenous rules developed through the PES implementation are far more likely to be respected and understood by local people (cf., Berkes, 2004; Ostrom, 1990) than externally-imposed rules (Cárdenas et al., 2000). This has been revealed in previous psychological studies (Deci and Ryan, 1985; Fehr and Falk, 2002; Decaro and Stokes, 2008). Likewise, Gong et al. (2010) stress the importance of trust among stakeholders in PES schemes which may outweigh the influence of economic incentives to achieve sufficient participation. Vatn (2010) builds further on these arguments and sees PES schemes contributing to the re-connection of decisions about land use management across different actors through cooperation. PES schemes achieve this in "a process mediated by existing institutions, which include property rights, legal frameworks, social perceptions and values" (Muradian et al., 2010). Consequently, Muradian et al. (2010) criticize that the "Coasean approach towards PES does not pay enough attention to the role of institutions and shared beliefs in shaping PES design and outcomes, even if these are critical under non-perfect market situations". Especially in the case of PHES, social perceptions about the relationship between land use and the provision of hydrological ecosystem services, although often not supported by hydrological evidence (Kosoy et al., 2007), play an important role. However, this can be a precautionary strategy to deal with uncertainty and incomplete information and does not necessarily represent a design drawback (Muradian et al., 2010).

In taking into account broader objectives beyond environmental conservation policy efficiency, the role of intermediaries may have to be reassessed. From a Coasean perspective intermediaries should be obsolete because market participants agree on their transactions on their own and additional players only raise transaction costs. However, when taking a broader perspective, intermediaries are often the key to effective performance of PES in practice. Kosoy and Corbera (2010) and also Vatn (2010) argue that intermediaries often take an active role in defining the services to be traded, setting the conditions among buyers and sellers, and largely influencing the price of the exchange.

Consequently, Muradian et al. (2010) argue that PES schemes, especially in developing countries, should not primarily be considered as “an economic tool only used to guarantee environmental protection in the most efficient way“ but instead should be seen explicitly in a broader context of development and equity goals with paying special attention to social embeddedness. Based on their arguments and in the context of uncertainty in the socio-ecological context of PES schemes and the required social embeddedness Muradian et al. (2010) propose an alternative definition of PES:

> *“as a transfer of resources between social actors, which aims to create incentives to align individual and / or collective land use decisions with the social interest in the management of natural resources.*”

This definition stresses the aim of ecological sustainability and the alignment with social goals which in the context of IWRM are as important as efficiency. Moreover, the definition leaves it open whether the transfer takes place in a kind of market context or through other mechanisms like incentives or public subsidies. According to Muradian et al. (2010), their definition encompasses the large diversity of existing PES which may be clustered according to three criteria: the importance of the economic incentive, the directness of the transfer and the degree of commodification of environmental services (see Table 5.2).

Criteria	**Explanation**	**Criteria rationale**
Importance of the economic incentive	Relative role of the payment in steering the desired land use of providers of ecosystem services	Multiple drivers may influence behavioral patterns
Directness of the transfer	Extent to which individual providers receive direct payments from the ultimate beneficiaries of the environmental service	Intermediaries play a critical role due to large coordination efforts between several suppliers and buyers
Degree of commodification	Extent and clarity with which compensation received by the ecosystem service providers has been defined as a tradable commodity	Characterization of the commodity is often fuzzy, based on assumptions about the relationship between land use and service provision

Table 5.2: Criteria for an alternative conceptualization of Payments for Ecosystem Services; based on Muradian et al. (2010)

The proposed clustering of PES schemes by Muradian et al. (2010) provides for different combinations of criteria while highlighting that scheme characteristics, in practice, go beyond the dichotomy between state-driven (i.e. rather Pigouvian) and private-driven (i.e. more Coasean). While more market-oriented schemes payments aim at increasing performance efficiency, they may also play an important role in facilitating the coordination between participants in schemes with a broader focus of objectives. Moreover, Muradian et al. (2010) identify two particularities of PES: the use of economic incentives and the high leverage of the intermediary in setting the rules. The setup of PES schemes generally implies more than defining the traded ecosystem service, reducing transaction costs and the allocation of property rights. A substantial degree of coordination between stakeholders, strategic decisions on trade-offs and dealing with information gaps is usually also part of it. Hence, in considering PES schemes as part of a policy mix, the PES scheme itself does not actually need to cover all costs if it provides synergies with other policy objectives. The coverage of at least the complete opportunity costs of alternative uses, for instance, may not be provided through a PES scheme, but still land users may be convinced through negotiation to accept less to bring themselves into compliance with commonly agreed rules for land use. Especially in developing country contexts where property rights are often unclear, land users may like to legitimize their land and compliance with an environmental regulation in the context of a PES scheme can present an opportunity to achieve this (Muradian et al., 2010).

Turning away from a strict Coasean approach has further implications. Muradian et al. (2010) stress that under Coasean efficiency considerations, the most appropriate PES strategy seems to be one which reduces the number and increases the scale of providers, simplifies practices, focuses on narrowly defined services, holds down transaction costs and complexity, while maximizing payment to reflect at least the opportunity costs of alternative land uses. Considering the broader potential of PES schemes Muradian et al. argue instead to apply PES “to develop local and regional institutional frameworks that can cope with complexity and diversity, and that can integrate PES within existing regimes of rural development and other policy instruments for environmental protection” (Muradian et al., 2010).

Tacconi (2012) proposes a further revised definition that builds on the definition by Muradian et al. (2010) but is more specific on three important characteristics of PES, namely transparency, additionality and conditionality:

> *“a PES scheme is a transparent system for the additional provision of environmental services through conditional payments to voluntary providers.“*

Transparency in PES schemes, defined as the timely and reliable provision of information to all relevant stakeholders (cf., Kolstad and Wiig, 2009), is important for the negotiation of contracts as well as for related provision and benefit valuation rules. Transparent ecosystem service valuation, negotiations and monitoring of compliance and service provision increases trust among potential stakeholders (Tacconi, 2012) and may avoid (perceptions of) corruption (Ferraro, 2008; Muradian et al., 2010). Furthermore, according to Mulgan (2000), in order to make accountable those who manage a system, i.e. here the PES scheme, transparency is needed. In contexts where parties who are in a weaker position in negotiating agreements are present, transparency is also important to provide much-needed information to these parties (Tacconi, 2012).

Additionality is another important characteristic and an equally difficult one to determine. PES schemes should of course lead to additional improvements in or at least maintenance of a desired status of ecosystem service provision compared to a hypothetical situation without a PES scheme. However, this hypothetical reference is generally difficult not to say impossible to define (Sommerville et al., 2009). Likewise it is often very hard to attribute ecosystem service provision to individual service providers and not to others. Thus, Tacconi (2012) argues that “additionality should be considered at the aggregate level for the whole PES scheme rather than for the individual ES providers“. However, in order to minimize the likelihood of wasting scarce resources additionality should be considered in the design of PES schemes (Tacconi, 2012).

The third characteristic of PES schemes is conditionality of payments (cf., Kroeger, 2012) which requires that service providers (sellers) only receive payment if they continuously comply with the contractually agreed-upon provision or flow of outputs (flow or output conditionality; (cf., Wunder, 2007; Engel et al., 2008)) or of inputs assumed to produce those outputs (proxy conditionality (cf., Quintero et al., 2009)). Input based payments, i.e. proxy conditionality are often related to specific agreements on land uses or land use practices which are put in relation to service provision. According to Kroeger (2012), different design options for PES schemes ranging from different forms of output (flow) to input (proxy) conditionality as well as different levels of strictness leave wide room for operationalizing conditionality in practice. Input conditionality, for instance, can incorporate reasonably fine-tuned payment levels for varying levels of the proxies (Murgueitio et al., 2004). However, if PES schemes employ conditionality, it is usually proxy conditionality (Wunder, 2007; Dillaha et al., 2007; Engel et al., 2008). Kroeger (2012) points out that examples of output conditionality are less common, primarily due to the difficulty and the cost of measuring actual outputs. Examples of output conditionality are documented by Wunder and Albán (2008); Honey-Rosés et al. (2009); Wendland et al. (2010).

These characteristics together define PES as a policy instrument that substantially differs from other, more conventional, instruments. The structuring in ecosystem service providers (or sellers) and buyers establishes new relationships based on the hydrological context and aims at defined environmental outcomes from which all stakeholders should benefit. The conditionality of PES in this context separates them from many other incentive-based resource management approaches. Conditionality is based on an agreement between the service buyer and the provider on a specified land use. Payments are only made if the agreed land use is complied with, thus, they are in some way subject to delivery of a quantifiable service, with specific terms and conditions often set out in a written agreement with the service provider (Engel et al., 2008; Porras et al., 2008). Flexibility can be introduced through agreements on different payments for different land uses, thus the payment may be scaled to expected performance. Finally, the voluntary nature of PES clearly distinguishes them from conventional command and control approaches. Thus, PES represent an alternative instrument to command and control approaches and other economic instruments (e.g. environmental taxes) in order to internalize environmental externalities. Moreover, because of their conditionality they are considered an advancement compared to ICDPs in the context of developing countries (Engel et al., 2008; Bond and Mayers, 2010).

The definition of PES proposed by Tacconi (2012) including the characteristics of transparency, additionality and conditionality applies to a variety of PES schemes at different geographical levels, from the international to the local level, including Coasean schemes involving individuals and businesses. However, the differences between types of PES schemes can be large and not all types are equally suitable to improve IWRM operationalization. The following section analyzes different Payments for Hydrological Ecosystem Services (PHES) characteristics in order to identify the most suitable type to promote fit and interplay as well as addressing operational constraints in the context of the IWRM implementation process.

The complexity of the conceptualization of PHES as more Coasean, Pigouvian or anything else demonstrates the existence of a broad variety of different PHES schemes in practice. Besides their economic conceptualization, PHES have been categorized with regard to their specific provider-buyer relationships, i.e. the characteristics of the service buyer specifically. In order to consider PHES as a complementary instrument to improve the operationalization of IWRM, specific PHES categories, i.e. types are more suitable than others. Thus, PHES types that borrow from the Coasean concept the directness of stakeholder, i.e. buyer and provider, interaction, and the ability to find context-specific solutions besides governmental regulation are more suitable for this purpose since participation and stakeholder involvement is not only desired but a prerequisite for effective outcomes of these schemes. Furthermore, scheme characteristics that are more associated with the Pigouvian conceptualization are also relevant for considering PHES schemes as an instrument in the context of IWRM. These characteristics include the involvement of the governmental actors as participants for the stewardship of social interests, provision of additional funding and connection to other policy levels in the sense of multi-level governance. Local municipalities, for instance, can facilitate the establishment of institutional settings and cross-jurisdictional cooperation in accordance with the principle of subsidiarity. Especially in the context of developing countries, strengthening of existing (formal) institutions is a legitimate objective for PHES. While governments may not be directly involved as a regulator in PHES schemes, they may still play an important role to facilitate institutional development based on existing institutions and implementation of the basin concept, e.g. land use / spatial planning based on of river basins.

The FAO acknowledges this variety of PHES schemes in a broad and general definition

> *"PES transactions refer to voluntary transactions where a service provider is paid by, or on behalf of, service beneficiaries for agricultural land, forest, coastal or marine management practices that are expected to result in continued or improved service provision beyond what would have been provided without the payment. The payment may be monetary or in some other form. PES transactions can involve a wide range of parties - including farmers, communities, taxpayers, consumers, corporations and governments - across a wide range of transaction types - from direct payments between downstream beneficiaries and upstream providers to consumers paying for a cup of "shade-grown" coffee beans produced on the other side of the world."*

Sommerville et al. (2009), for instance, argue that there are a wide variety of situations where ecosystem service suppliers and buyers can operate. These situations may be characterized through a continuum of dichotomies of the service providers, the buyers and the relationship between these two as summarized in Table 5.3.

Characteristics	Institutional context		Options[1]	
Service provider	governance type	democratic	⟷	authoritarian
	type of provider	individual	⟷	community
	property tenure	private property	⟷	no tenure
	legality of behaviors	legal	⟷	illegal
	opportunity costs	homogenous	⟷	variable
Service buyer	buyer's funding	secure	⟷	insecure
	buyer goals to trade-off	economic efficiency	⟷	equitable distribution
	additional buyer goals to trade-off	social	⟷	ecological
Relationship	threats to system	internal	⟷	external
	distance between buyer and provider	local	⟷	international intermediaries
	buyer-provider relationship	one-on-one	⟷	one-off negotiation
	negotiations	market-based	⟷	regulated
	participation constraints	voluntary	⟷	service

Table 5.3: Possible institutional contexts of PHES implementation depending on characteristics of service providers, buyers and their relationship; based on Sommerville et al. (2009)

[1] Options on the left often represent easier contexts for the implementation of PHES. There is often a continuum of options from left to right.

Four principal types of PHES schemes may be distinguished based on the provider and buyer constellation (from local to national and direct to indirect), i.e. buyer characteristics:

(1) **User-financed** (Asquith and Wunder, 2008), sometimes also referred to as self-organized (private) deals (Johnson et al., 2001; Smith et al., 2006; Greiber and Schiele, 2011), bilateral agreements and beneficiary pays funds (Bennett et al., 2013) or Voluntary Contractual Agreements (Tognetti et al., 2006). These schemes are of local or regional character and "typically involve the negotiation and agreement of a contract in which resource users, who benefit from watershed services, compensate upstream landowners for the costs of adopting management actions needed to insure provision. Intermediary organizations, such as landowner associations, are often necessary as a way to reduce transaction costs associated with the need for agreement and collaboration among; numerous downstream beneficiaries and landowners dispersed over large upper watershed areas" (Tognetti et al., 2006).

(2) **Government-financed** (Asquith and Wunder, 2008) or public-payment-schemes (Johnson et al., 2001; Smith et al., 2006; Greiber and Schiele, 2011). These are often national programs where the federal government pays on behalf of the society as the beneficiary.

(3) **Trading schemes** (Johnson et al., 2001; Smith et al., 2006; Greiber and Schiele, 2011), where a market for pollution rights is established.

(4) **Certification and labeling for products** that through their production promote the provision of ecosystem services (Tognetti et al., 2006).

PHES designed as user-financed, self-organized deals or voluntary contractual agreements are specifically interesting in the context of the operationalization of IWRM since they imply direct interactions of hydrological ecosystem service providers and buyers taking into account the specific context of natural and human system interaction. Moreover, they are often realized either as small scale local initiatives but may also take the form of larger water funds with significant budgets for investments, e.g. the Quito Water Fund. Goldman-Benner et al. (2012) define a water fund "as a PES approach that for financial management uses a trust fund managed by an external entity. In addition, water funds share the following criteria: (1) multiple water service users or user groups, (2) payments that support implementation of watershed best-management practices and conservation, and (3) a board of directors with stakeholder representation that decides how to spend the revenue". According to Goldman-Benner et al. (2012), the funds are governed by a multi-institutional body, i.e. a public-private partnership that includes service buyers, and in some cases sellers, which makes decisions about how to spend water-fund revenue. Thus, water funds can be understood as an organizationally and financially more advanced form of local PHES schemes. Water funds have been introduced mainly in Ecuador, Colombia and Peru.

Considering the growing popularity of PES in general and PHES specifically, several global and regional reviews (see Table 5.4) have been realized over the past decade in order to take stock of their distribution and number among different countries as well as their relevance and trends in implementation.

Besides providing general information on the development of the instrument, these reviews reflect also the general trend to shift from a strict Coasean market perspective on PHES towards a broader interpretation. The first reviews of Perrot-Maître and Davis (2001); Johnson et al. (2001) and Landell-Mills and Porras (2002), for instance, still referred to Markets for Ecosystem Services, thus, stressing a Coasean conceptualization. In contrast, the following reviews more or less discarded this term and used Payments for Ecosystem Services instead. Porras et al. (2008), who also authored the review of 2002 (Landell-Mills and Porras, 2002), made this explicit based on the prevailing evidence of the actual nature of PES implementation in practice. The shift from focusing mainly on forest ecosystems in the first reviews (Johnson et al., 2001; Perrot-Maître and Davis, 2001; Landell-Mills and Porras, 2002; Dudley and Stolton, 2003) as provisioning ecosystems of hydrological ecosystem services to a broader consideration of other ecosystems, including agro-ecosystems, is also a trend that can be observed as a result of PHES evolution over time. This documents an increasing broadness in the application of the instrument in different environmental policy contexts. However, several reviews on the importance of protected forests for the provision of drinking water have made a special business case for PHES (Dudley and Stolton, 2003; Ernst, 2004; Postel and Thompson, 2005; Stanton et al., 2010; Buric et al., 2011; Bennett et al., 2013). Hence, for many single and large service buyers, e.g. water companies of large cities or drinking water bottlers, PHES has already become a cost-effective instrument. For instance, the City of New York started a PHES scheme in the 1980s to solve water quality problems in the Catskill and Delaware river basins. This case provided a high profile and very tangible example of their business case potential (NRC, 2000; Appleton, 2002; Pires, 2004). Another example is a PHES scheme by Nestlé Waters to secure the quality of its Perrier Mineral Water. This case served to confirmed that PHES could be both successfully implemented and highly cost effective for the buyer (cf., Perrot-Maître, 2006; Bond and Mayers, 2010). In the United States of America (US), many medium- and large-sized cities followed the example of New York and almost all water quality trading schemes are located in the US as well (Ernst, 2004; Bennett et al., 2013). In developing countries, there are also a number of PHES schemes where a single beneficiary, usually a company which uses water as direct input to their production process, is directly contracting land users for the provision of hydrological ecosystem services. A typical case of this kind of PHES is the La Esperanza hydro power PHES in Costa Rica (Rojas and Aylward, 2002; Porras et al., 2010). These schemes are characterized by a rather marginal role or even absent intermediaries and represent

Year	Scope and subject of the case study review	Source
	Global reviews	
2001	9 examples from developing and developed countries focusing on different characteristics depending on the financial mechanism.	Johnson et al. (2001); Perrot-Maître and Davis (2001)
2002	287 cases of MES (61 for Watershed Protection) from developed and developing countries. Focus on different market forms for different ecosystem services.	Landell-Mills and Porras (2002)
2003	105 of the largest cities in developing and developed countries. Focus on the importance of protected forests for the provision of drinking water.	Dudley and Stolton (2003)
2004	25 PES (21 local ones and 15 for watershed protection) cases from 15 countries in the Western Hemisphere. Focus on main differences and similarities between different PES, as well as their strengths and limitations.	Mayrand and Paquin (2004)
2008	81 case profiles of PHES from developing countries. Focus on national and local schemes addressing an externality, voluntariness on the provider's side and conditionality on the agreed land use.	Porras et al. (2008)
2008	Comparative case study comparison of 14 PES schemes from developing and developed countries. Focus on PES schemes closer to complying with the Coasean conceptualization.	Wunder et al. (2008)
2010	Multi-country analysis of 10 PHES schemes over three years of implementation in developing countries. Additional review of 50 PHES cases from developing countries. Focus on scheme characteristics and impact on livelihoods and environmental outcomes.	Bond and Mayers (2010)
2010	127 PHES and water quality trading schemes from developing and developed countries. Focus on the full account of payments directed to protect or restore hydrological ecosystem services.	Stanton et al. (2010)
2011	36 PHES schemes for cities from developing and developed countries. Focus on establishing a global inventory of cases and their characteristics.	Buric et al. (2011)
2011	Meta-analysis of institutional-economic factors of 47 PHES schemes from developing countries explaining their environmental performance.	Brouwer et al. (2011)
2012	163 PHES schemes from 34 developing countries. Focus on basic features and governance processes.	Lin and Nakamura (2012)
2013	205 PHES and water quality trading schemes from developing and developed countries. Focus on profiling the scale, size, shape, and direction of investments.	Bennett et al. (2013)
	Latin American reviews	
2007	90 case studies with PHES characteristics. Review on characteristics of initiatives.	Southgate and Wunder (2007, 2009)
2012	38 PHES schemes. Systematical analysis to understand key features and to identify information needs for evidence-based policy design and implementation.	Martin-Ortega et al. (2012)
	Central American reviews	
2002	31 PES schemes (11 PHES). Focus on implementation process with respect to legal, financial, organizational and institutional aspects of different countries.	Mejías Esquivel and Segura Bonilla (2002)
2007	Comparative analysis of 3 local PHES case studies. Focus on socioeconomic background, opportunity costs and stakeholders' perceptions of the conditions of water resources.	Kosoy et al. (2007)
2007	4 local PES schemes (2 PHES). Focus on equity consideration of scheme participants in protected areas compared to rural communities.	Corbera et al. (2007)
2008	27 PHES schemes mostly of local scale in response to problematic water supply.	FAO-FACILITY (2008)
2008	3 local PHES schemes. Focus on functioning, actors involved, achievements + limitations.	Martínez Tuna (2008)

Table 5.4: Overview of reviews on PHES schemes at different geographic scales used to derive typical features of the instrument and its implementation

a compelling business case where a buyer of a service looks for a cost-efficient solution to a given externality problem. Besides these purely private PHES schemes, the large majority of local PHES involve local governments and / or other public authorities, often in form of public municipal water suppliers as beneficiaries as well as important intermediaries for the implementation and execution of schemes.

Porras et al. (2008) describe the evolution of *first generation* followed by *second generation* schemes as a notable development of the instrument over the last decade. While the *first generation* were characterized as relatively isolated pilot schemes in form of learning by doing approaches - many of them are documented by Landell-Mills and Porras (2002), the *second generation* schemes are often already taking into account existing experiences and lessons learned from the former projects. Consequently, these schemes place stronger emphasis on the design of baseline studies, monitoring and information sharing which have been identified as particular problems of first generation schemes (Porras et al., 2008). Hence, hydrological measurements and valuation studies are more often part of the instrument design of *second generation* schemes. Additionally, earlier (first generation) schemes have been modified in order to promote the participation of small farmers and respond more to stakeholder concerns, e.g. the national program in Costa Rica, or in order to improve the analysis of hydrological linkages as in the case of Los Negros, Bolivia (Porras et al., 2008). Besides an increasing recognition of the need for site-level measurement due to the complexities of the land use and water linkages, Porras et al. (2008) stress that schemes are diversifying potential areas of service provision. Thus, the contribution of other types of land-use such as agroforestry and organic agriculture is becoming more recognized. This results in additional stakeholders entering on the supply side, e.g. farmers complementary to owners of forest, and in broadening the range of potential intermediaries and facilitating organizations, e.g. forest and conservation organizations have been joined by organizations promoting sustainable agriculture, agroforestry and community development. Many of these *second generation* schemes are initiated as part of larger regional projects such as Cuencas Andinas of the German Agency for International Cooperation (GIZ), the Silvopastoral Project of the Swiss Agency for Development and Cooperation (SDC) or the Water Funds of The Nature Conservancy (TNC). Increasingly national programs are developed, often triggered by good experiences with local pilot schemes. There are also more and more PHES schemes that are financed additionally through national programs or funds complementary to service buyer investments. Bond and Mayers (2010) point out in their global review and multi-country analysis of PHES that payments "should contribute to the costs of watershed management and, if upstream communities are also characterised by poverty, these payments should contribute to local development and poverty reduction as well". This again marks a development from considering PHES as only buyer, i.e. service provision oriented towards contributing to the financing of a more general objective of water management at the river basin level.

In 2008, Porras et al. presented a follow up report of the review of Landell-Mills and Porras (2002), now with a focus on hydrological, i.e. watershed services. The number of PHES schemes increased from 41 in 2002 to 102 ongoing cases in 2008. In the end, sufficient information was gathered on 81 cases to be further assessed in the study of Porras et al. (2008)[5]. Besides stressing that most PHES are located in Latin America (primarily in Ecuador, Colombia, Bolivia, and almost all the countries in Central America except Belize), they also highlight a remarkable growth in the number of schemes and proposals particularly in Latin America followed by Asia. This is confirmed by the reviews of Southgate and Wunder (2007, 2009) from USAID and Virginia Tech based on Dillaha et al. (2007). They identified 90 PHES case studies at different stages of implementation, about half of them in South America and the other half in Central America and the Caribbean. Most of these cases are "PHES-like", meaning, that they are of local character but without a strong market mechanism. Similarly, Porras et al. (2008) identified that most schemes are local in nature, operating at the level of small river basins. The share of local initiatives at the level of river basins as opposed to national programs has increased from 68 % in 2002 to 82 % in 2008 (Landell-Mills and Porras, 2002; Porras et al., 2008). Brouwer et al. (2011) carried out a meta-analysis of 47 PHES schemes for which sufficient information was available. The authors determine the average age of the schemes by the year 2010 to be 11 years, almost 70 % of the schemes are located in Latin America, with two thirds of them located in Central America. Another recent global review of Lin and Nakamura (2012) identified 163 PHES schemes in developing countries at different states of implementation (9 schemes have been abandoned). Again, the large majority of PHES schemes (82) are located in Latin America, 42 in Asia and 39 in Africa. 136 of the PHES schemes are local ones, 27 represent national programs. Hence, it can be concluded that the largest number of PHES, especially local ones, and the longest experience with the application of the instrument exists in Latin America (Landell-Mills and Porras, 2002; Dillaha et al., 2007; Porras et al., 2008; Lin and Nakamura, 2012).

Although the number of PHES schemes characterized as local is growing, they are often not completely independent from national schemes or national government's assistance. A relationship between national and local PHES schemes was identified by Porras et al. (2008) in half of the cases they examined (41 out of 81). This relationship is mostly in form of financial and/or technical assistance from the national level to the local initiatives in establishing negotiations among stakeholders, preparation of baseline studies, design of mechanisms for collecting and allocating payments and general management of the scheme. In some cases additional contributions from national funds complement those collected at the

[5] Updated case study profiles are available at: www.watershedmarkets.org

local level (Porras et al., 2008). However, according to Porras et al. (2008) different types of links between the national and local level exist:

- National programs support pre-existing local schemes. For example in Valle de Bravo in Mexico, funding from the national Payments for Hydrological Environmental Services program was used to supplement the existing voluntary contributions to an environmental fund.
- National programs lead to the creation of local schemes. For example in Costa Rica, the existence of the national-level PES program has provided the framework and institutional capacity to spur local-level agreements with several hydroelectric companies.
- Local schemes lead to the creation of national programs. For example the small local schemes coordinated by the Swiss Agency for Development and Cooperation (SDC) / Program for Sustainable Agriculture on the Hillsides of Central America (PASOLAC) in Nicaragua, El Salvador and Honduras are helping to create the momentum for the creation of national-level programs.
- Local schemes set up as a pilot for national programs as in the case of Coatepeque and Jaltepeque-Jiquilisco in El Salvador, where the planned national program *Ecoservicios* is being piloted.

The other half of the local schemes examined by Porras et al. (2008) emerged independently of a regional or national program.

An interesting property of PHES schemes in the context of IWRM implementation are the multi-level governance links. Apparently, local schemes can create a somehow *upward connectivity* to higher governmental levels enabling institutional interplay for financial and / or technical assistance in establishing negotiations among stakeholders, preparation of baseline studies, design of mechanisms for collecting and allocating payments and general management of the scheme. These relationships may work both ways, local initiatives resort to expertise of higher level public authorities and these may learn from local contexts and knowledge. The co-financing and -management can represent a benefit to the different levels as well. The financing of start-up costs of local initiatives is often covered by international donors, directly or through a national NGO, reoccurring costs and then mostly covered through payments by water users (Porras et al., 2008). Moreover, local or national governments often contribute to additional ongoing financing. For example, local schemes in Costa Rica involving hydroelectric companies or user fees from domestic water users, as in the case San Pedro del Norte in Nicaragua, combine funding from local government budgets with private-sector or individual user investment (Porras et al., 2008). Furthermore, national and local PHES schemes can be complementary in their target areas. Porras et al. (2008) found out, for instance, that while local schemes often focus on improved land use practices at agricultural sites (e.g. agroforestry, soil conservation), national schemes tend to focus stronger on the conservation of existing forests or reforestation. However, local schemes with focus on improved land use practices often attempt to generate medium-term to long-term on-site returns for the farmer as well. The higher flexibility of local schemes reaches beyond the consideration of more diverse land uses for service provision, because monitoring and adapting to local conditions is easier, the inclusion of various types of land tenure is also more common in local schemes (Porras et al., 2008). Based on their global review of PHES cases, Stanton et al. (2010) consider those PHES schemes that “adopt an all-inclusive approach in terms of identifying the stakeholders; those that have been able to evolve over time and adapt for more effective and comprehensive resource management, and those that have been flexible enough to take advantage of linkages between local, private bilateral programs, and the larger national incentive programs“ as most successful.

The report by Stanton et al. (2010) represents an attempt to catalog the use of PHES across the world and the financial resources that are spent for this purpose. A major insight of this report is that the national PHES programs of China, Mexico, Costa Rica, and the US (see Table 5.5; national PHES programs exist as well in South Africa, China and the Philippines) form, in monetary terms, the largest group of PHES schemes but the least innovative interventions. This is because they are often more or less a different way of using governmental resources and tax revenues to pay for conservation. Nevertheless, these national schemes create a demand for hydrological ecosystem services which seems to provide incentives for conservation (Stanton et al., 2010). While the American and Chinese national programs are the largest, the emerging leader in terms of experimentation with government payments for hydrological ecosystem services, according to the authors, is Latin America. New PHES designs, both in terms of how the payments are made, as well as in how their effects are measured, monitored, perfected, and replicated are experimented with mainly in Latin America. The use of trust funds to engage public and private sector resources, for instance, is a particular Latin American innovation.

Martin-Ortega et al. (2012) reviewed 38 different PHES schemes in 10 Latin American countries and analyzed four principal characteristics: (i) context of the scheme (location, spatial scale and year of implementation); (ii) stakeholders (different parties involved for (1) implementation and management in form of initiators and intermediaries; and (2) participants in form of sellers and buyers of services); (iii) targeted service and actions; and (iv) contract details (land use specified, duration, payment amounts and type, valuation technique). In accordance with previous global and regional PHES reviews, Martin-Ortega et al. (2012) identified a larger number of schemes implemented at the local level (74 %) and about 18 % having at least a locally implemented component but are considered as partly national schemes. The remaining schemes are considered national programs. Of the 42 % of the cases where one or more environmental threats

Country	Program name	Year	Description
Colombia	Plan Verde	1999	National governmental forestry plan aiming at recovering forest cover while protecting micro-watersheds, regenerating areas affected by forest fires and degraded mangroves. Driven by the government's recognition of the need to protect the ecosystems that influence hydroelectricity production, drinking water supply and irrigation.
Costa Rica	Programa de Pago por Servicios Ambientales	1997	Government-led scheme rewarding forest owners for protection of water, carbon sequestration, biodiversity protection and landscape beauty from forests. Most of the funding still relies on state funds derived from a fuel and water tax, with increasing participation from the private sector (especially hydroelectric projects).
Mexico	Programa para el Pago por Servicios Ambientales Hidrológicos	2002	Country-wide scheme targeting areas of well-preserved natural forest for protection of their hydrological function in critical watersheds and over-exploited aquifers and proximity to water sources that supply settlements of more than 5,000 inhabitants, which might in the future take over the payment through their own local government and/or water utilities.

Table 5.5: National programs of Payments for Hydrological Ecosystem Services in Latin America; from Porras et al. (2008)

were specified within the objective of the schemes, deforestation and land cover loss was by far considered the biggest threat (77 %), followed by water pollution (32 %) and water over exploitation (23 %). Cattle expansion (10 %) and other threats (14 %) including lack of water treatment facilities or lack of access to water or sanitation and forest fires were mentioned as well.

The numerous reviews on PHES over the last decade document both a growth in number and a growth in investment in this instrument. A large number of local schemes and several national programs have been established. After more than a decade of experience with the instrument, the number of applications is still increasing while the instrument is applied in a broader context and continuously improved. Although only PHES schemes where local governments play a role will be considered in the following assessment, in the context of IWRM operationalization, the preliminary conclusion that PHES are able to engage a broad variety of public and private beneficiaries from hydrological ecosystem services can be drawn. However, the longest and most wide-spread experience with the application of PHES exists in Latin America. While in South America a trend towards schemes in form of water funds can be observed, there are many applications of local schemes in Central America. Because of their wide-spread popularity, PHES schemes of Latin America, and especially Central America, have been subject to specific reviews and peer-reviewed publications in form of comparative assessments (see Table 5.4). Based on these comparative assessments the principal implementation steps of PHES schemes are described and analyzed with respect to a potential contribution to the operationalization of IWRM in the following.

5.2.1 Typical implementation process of PHES

In order to identify potential contributions to the operationalization of IWRM, the general implementation process of locally organized and at least partly user-financed voluntary contractual agreements on Payments for Hydrological Ecosystem Services is documented in this section. Contrary to traditional policy instruments (e.g. command and control or environmental taxes), the implementation process of these local PHES schemes is characterized by a bottom-up procedure with different actors participating in the process of instrument design and implementation. However, there is always some influence from higher governance level as they provide the general legal, institutional and economic context. The typical setup of local PHES is illustrated in Figure 5.3.

The most common point of departure for the development of a PHES schemes is the perception of degradation of water resources, often associated with land use changes (e.g. deforestation or increasing agricultural activity), and often also disappointment with past measures that failed to address environmental degradation adequately. Although there are cases were the idea of a PHES scheme comes from the supply side, e.g. in the case of protected forests which are endangered by deforestation, it has been repeatedly stressed that demand driven PHES are more promising since beneficiaries are aware of an environmental problem and do not have to be "convinced" of it. However, according to Landell-Mills and Porras (2002), most (early) PHES schemes have emerged as a result of growing willingness to pay among beneficiaries, most often related to an improved understanding of the benefits provided by watershed and the increasing threats to these.

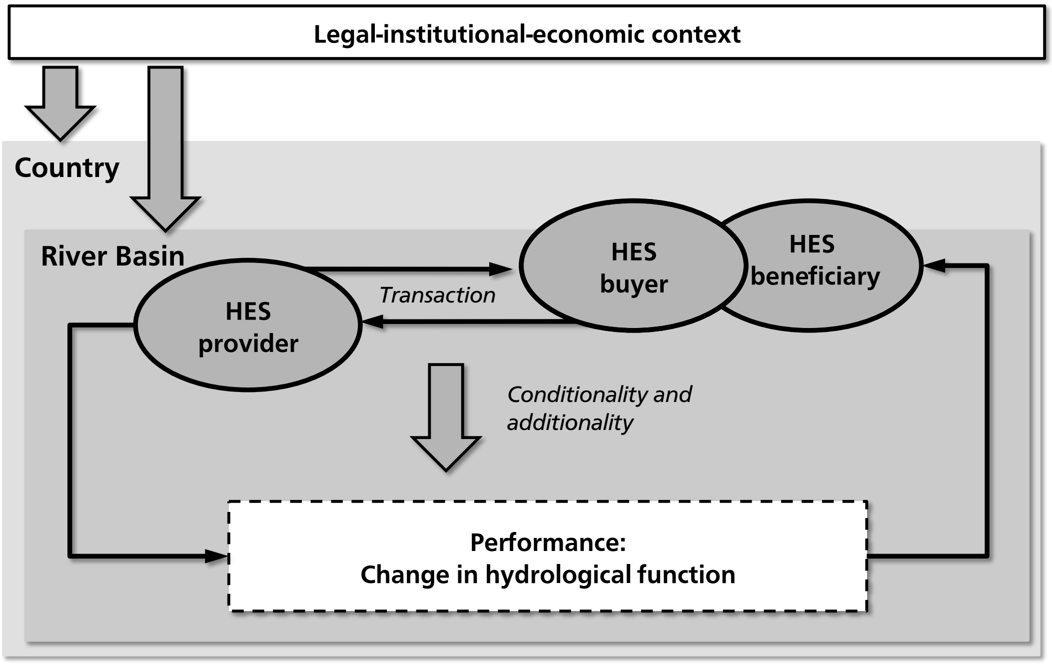

Figure 5.3: Conceptual framework of Payments for Hydrological Ecosystem Services; from Brouwer et al. (2011)

In developing countries PHES often represent a response to failed governmental regulation, while in more developed countries, new government regulations for improved water quality have been identified as a significant force behind investment (Landell-Mills and Porras, 2002).

According to Hedden-Dunkhorst et al. (2010), in many cases intermediaries such as governmental and non-governmental organizations, research teams, or consultancies play a vital role in initiating and subsequently implementing a scheme based on their skills and expertise. The role and importance of different actors and intermediaries will be discussed in Section 5.2.2.

In general, the establishment of a PHES schemes require several technical and coordinating steps (see Figure 5.4) which ideally should be carried out in order to become a successful scheme (cf. Engel et al., 2008; Tacconi, 2012). The implementation process provides a number of synergies with the requirements for IWRM operationalization. Even when PHES implementation is carried out without considering potential linkages to the IWRM operationalization, it still yields results that are beneficial to the IWRM process. As mentioned before a prevailing feature of the PHES implementation process is that solving the problems of fit and interplay is an inherent part of it. Moreover, several common operational constraints of IWRM may also be overcome. The general steps of PHES implementation are presented in the following.

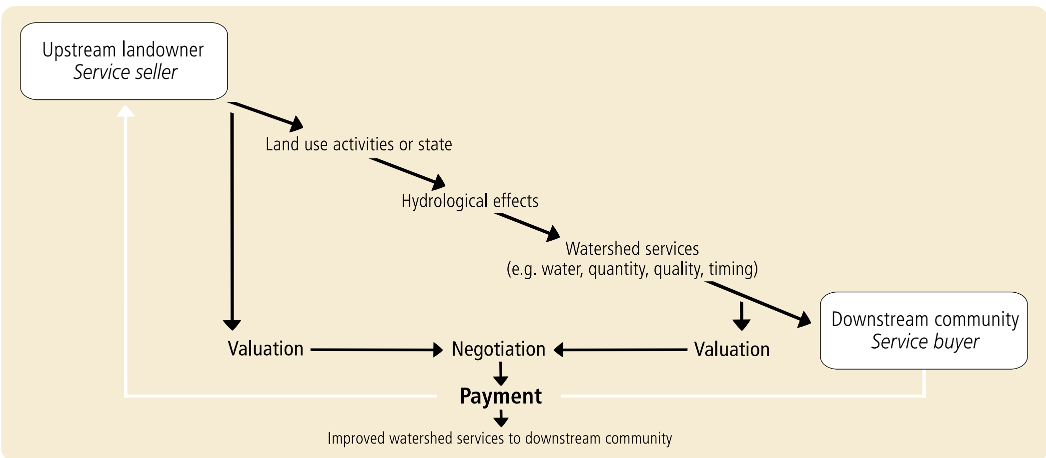

Figure 5.4: Payments link upstream and downstream stakeholders in watershed services (Smith et al., 2006)

Four principal steps towards PHES implementation can be distinguished and grouped into a preparatory and an instrument application and execution phase:

- **Preparatory phase**
  - Bio-physical cause-effect relationships between land use and service provision
  - Socio-economic valuation of provision of and benefits from ecosystem services
- **Instrument application and execution phase**
  - Negotiation between stakeholders towards agreements on land use
  - Continuous realization of agreed actions, payments and monitoring

The first two steps represent an application of the hydrological ecosystem service concept for a given river basin context. These steps identify, in a preparatory manner, the appropriate scale for management, i.e. application of the PHES scheme, by addressing the issues of fit and interplay. The last two steps form the actual application of the instrument in form of reoccurring events. The individual implementation steps will be described and analyzed in the following with regard to the extent they address the operational constraints identified in Section 3.3 and how the instrument provides complementary incentives towards IWRM operationalization as part of a policy mix.

The very first step of implementation represents the bio-physical assessment. This implies the identification of potential areas for the provision of hydrological ecosystem services and potential beneficiaries of these services. As described in Section 4.4 the logic of the ecosystem service approach here addresses the problem of fit, as defined by prevailing ecosystem properties and identified cause-effect relationships, by taking into account the institutional interplay with potential beneficiaries. Research institutes or universities are often engaged to carry out a biophysical assessment of the current state of the water resources and try to identify potential areas of service provision. This way an appropriate size for a management unit is identified based on hydrological linkages between provisioning areas and beneficiaries (Tacconi, 2012). It is no surprise that hydrological ecosystem services payment schemes are local, often involving watersheds that supply urban or rural settlements in their proximity. Hydrological linkages between upstream actions and downstream water impacts are less verifiable with increasing size of river basins and perceived links by beneficiaries and suppliers are less likely (Landell-Mills and Porras, 2002). Hence, unless downstream beneficiaries believe that they will receive sufficient gain from upstream watershed protection, they may not be willing to pay. Typical spatial scales of PHES implementation identified by Brouwer et al. (2011) in a review of 47 schemes are illustrated in Figure 5.5. Hence, a typical spatial scale of provider and buyer interaction is in almost 50 % of the cases reviewed by Brouwer et al. (2011) smaller than 1000 ha.

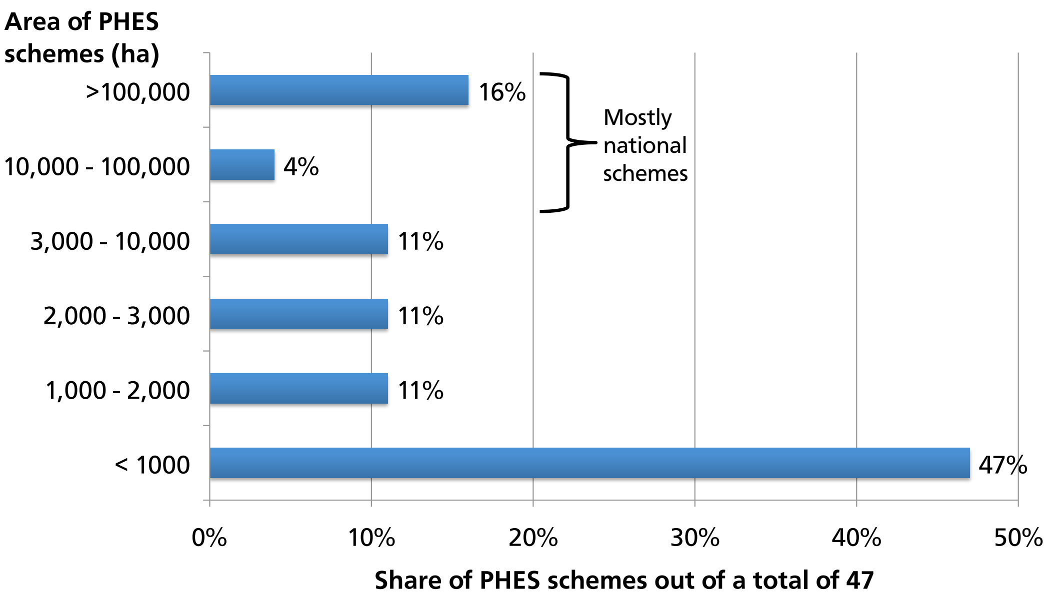

Figure 5.5: Identified scales of operation of PHES schemes; from Brouwer et al. (2011)

The geographical localization of provisioning areas and beneficiaries provides the baseline scenario which is aimed to be maintained, in case of an precautionary objective, or to be improved when degradation has already occurred. For this baseline scenario spatially explicit identification of different land uses and the status of water resources is needed. If historic information is also available, e.g. land use information or hydrological information, a deteriorating trend may be identified. Often these are the first data acquisitions at the scale of the considered river basins. Hence, if this baseline scenario is carefully determined, very useful information (e.g. on land uses, soils and their properties, water quality and

Benefits from HES provision	Hydrological Ecosystem Service (output)	Possible provision proxy (input)
Avoided drinking water filtration costs (sediment) Avoided dredging costs / reduction of reservoir lifetime Avoided water filtration cost for industrial users (sediment) Avoided dredging costs / loss in agricultural productivity	Reduced sediment load in drinking water (municipal water plant intakes) Reduced sediment input in hydro reservoirs Reduced sediment load at industrial water intake points Reduced sedimentation of irrigation systems	Riparian vegetation buffers; intact natural land cover; intact floodplain; undisturbed river channel
Avoided drinking water treatment costs (livestock bacteria)	Reduced bacterial load (fecal coliform, fecal enterococci) in drinking water at municipal water intake	Riparian buffers or forests
Water-based recreation	Water quantity and quality (suspended sediment, bacteria, nutrients) in surface waters used for recreation	Riparian buffers or forests; natural land cover; intact upstream vegetation (water holding capacity)
Avoided damages to human life, health, property Avoided drinking water shortages Avoided industrial and agricultural water shortages Avoided reductions in electricity generation	Avoidance of flood water levels in water bodies in populated and cultivated areas Drinking water availability Water availability for industrial and agricultural users Water availability for hydro power generation	Functional headwater and floodplain wetlands and forests (soil water holding capacity; vegetation reducing runoff)

Table 5.6: Specific benefits from hydrological ecosystem services and related proxies for provision; based on Kroeger (2012)

quantity) are gathered which are often lacking for IWRM operationalization (referred to as information gap in Section 3.3). The studies on hydrological function and the impact of human induced land use changes of the páramo ecosystems in the Andes by Buytaert et al. (2005; 2006; 2007), for instance, built the basis for the development of water funds (Goldman-Benner et al., 2012). Although hydrological studies in this complex form are an exception, survey on land uses, soil properties and basic hydrological parameters (e.g. precipitation, climate variables, river discharge) are carried out in many cases in order to build a theory of hydrological ecosystem service provision and identification of important provisioning areas. Specific benefits from hydrological ecosystem services and potentially related proxies for provision are summarized in Table 5.6.

In a next step a socio-economic valuation of potential hydrological ecosystem services to be provided is done, often carried out either by NGOs or university and research institutes. This is the first step to prove the feasibility of a PHES by identifying potential service providers and service buyers as well as their respective opportunity, replacement, restoration or avoided costs. Although the different stakeholders involved in this process are usually not in direct contact at this phase of implementation, they may already get a perception of how service provision is valued by different counterparts' acceptance or willingness to pay. In some cases, the bio-physical assessment and the economic valuation of identified hydrological ecosystem services is carried out together in form of scientific studies. At best the biophysical assessment provides scientific evidence of certain land use effects on the provision of hydrological ecosystem services for the considered river basin which serve as an argumentation basis to engage service buyers and providers. This is rarely the case in practice, instead, cause-effect relationships between land use and alternation of hydrological functions (on-site) form the basis for assumed service provision. Moreover, in many cases the cause-effect relationships are instead based on shared beliefs, for instance with respect to a general role of forests for the provision of services (Engel et al., 2008; Muradian et al., 2010). However, the establishment and communication of cause-effect relationships between land use and service provision is required to engage potential service providers and, more importantly, service buyers. The awareness that certain land uses may maintain or improve service provision to beneficiaries is a requirement for estimates of willingness to pay for them (Landell-Mills and Porras, 2002). For at least 9 out of 38 of the PHES schemes reviewed by Martin-Ortega et al. (2012) a Willingness To Pay (WTP) study was documented, technical studies as biophysical assessments were reported for 11 schemes. However, their review did not find information on opportunity costs studies.

Particularly favorable conditions for implementing a PHES scheme are present when potential upstream provisioning areas are characterized by low opportunity costs (e.g. marginal agricultural land), basic hydrological cause-effect understanding exists and downstream water users are able to compensate for service provision (e.g. larger urban areas,

Promoted land use	Example	Reference
Conservation and protection of existing ecosystems	In Costa Rica most PES is allocated per hectare of land protected each year	Russo and Candela (2006); Sánchez-Azofeifa et al. (2007)
	Voluntary forest conservation contracts in Norway	Barton (2010)
Agricultural practices aimed at providing environmental services and on-site economic returns for farmers	Agroforestry contracts in the PSA Programme in Costa Rica, Sumberjaya in Indonesia, and Jesus de Otoro in Honduras	Corbera et al. (2007)
	Silvo-pastoral projects in Colombia, Nicaragua and Costa Rica	Pagiola et al. (2007)
	Best management contracts in the Catskill-Delaware Watershed in New York	Appleton (2002)
Reforestation for commercial purposes	Six national PES schemes and approximately 11 small local watershed schemes promote reforestation	Porras et al. (2008)
Rehabilitation of degraded ecosystems for protection	Removal of alien tree species, Working for Water in South Africa	Binns et al. (2001)
	PCJ in Brazil to restore riparian forests	Porras et al. (2008)

Table 5.7: Examples of promoted land uses in PHES schemes; based on Porras et al. (2013)

commercial agriculture, industry, hydro power). However, as it is a complex task to link land use actions in a spatial and temporal manner to specific outcomes, Tognetti et al. (2006) stress that making uncertainty explicit may be critical for managing buyer expectations and maintaining their cooperation in the long term. Agreements on how to deal with uncertainty is also critical in negotiating an equitable distribution of costs and benefits between service providers and beneficiaries.

In the preparatory phase of PHES implementation intermediaries play a crucial role in identifying supply and demand of hydrological ecosystem services. Although NGOs and international agencies are often leading this phase, there are also actors from the supply and / or demand side involved with a principal interest in improving the service provision in the future. In two thirds of the reviewed schemes Martin-Ortega et al. (2012) identified various driving actors, in more than 40 % of the cases a national or local NGO as the leading driver at the origin of the scheme followed by municipalities and governments or governmental agencies (23.7 and 18.4 % respectively). In around 16 % of the schemes participants themselves, i.e. buyers and sellers, are among the promoters (Martin-Ortega et al., 2012). They assume that when different entities, with different capacities and knowledge, are involved, the program is more likely to get implemented. There is an inherent motivation to gather as much information as feasible about the potential demand and supply side. As a result, on the one hand, a process of land use categorization (at least in areas of high or low potential for service provision) within the area of hydrological influence takes place in order to demonstrate possibilities for improvements in service provision to potential service buyers. On the other hand all water users and potential beneficiaries of hydrological ecosystem services are identified to create the demand side. Thus, already in this preparatory phase the problems of fit and interplay are addressed to find the right scale for the implementation of the instrument, driven by inherent incentives of the instruments implementation logic to do so. The specific biophysical and socio-economic context make up the framework for this endeavor. At this phase of the implementation process all potential stakeholders on the supply and demand side may be addressed across sectors, e.g. different land use sectors such as agriculture, forestry, nature reserves as well as different water use sectors such as irrigation agriculture, water using industry, hydro power generators, water supply companies and fisheries. Since this process is based on the principles of the ecosystem service concept and not lead by a regulator with respective administrative boundaries, an appropriate fit is achieved through the identified ecosystem cause-effect properties.

Porras et al. (2008) and also Martin-Ortega et al. (2012) found out that forest conservation and management as well as reforestation are most often associated with improved hydrological service provision, but increasingly general watershed conservation / restoration and improved agricultural practices (e.g. agroforestry) are recognized to be potentially beneficial as well. Examples of typically promoted land uses in PHES schemes are listed in Table 5.7.

Following the bio-physical and socio-economic assessments the actual process of negotiation is carried out. Again intermediaries play an important role in this phase in order to involve all relevant stakeholders as well as in moderating the process. On the supply side, providers of hydrological ecosystem services have to be engaged. This requires that they are legitimated to manage the targeted land (either through existing formal property rights or informal established rights to use the land) and are willing to provide the service. The definition of services provided is often based on land use proxies, hence, the size of different provisioning areas has to be determined and an agreement on the land use proxy has to be reached. According to the findings of Engel et al. (2008) and Martin-Ortega et al. (2012), in general the payment is determined directly by an action (input related) and not by the results of the action on the actual hydrological ecosystem service (output related). Martin-Ortega et al. (2012) revealed that about three quarters of the PHES schemes they reviewed include a bundle of ecosystem services, and about half of them include not only water related services, but also other types of services (e.g. carbon sequestration). However, a very large majority of the schemes focuses on improving extractive water supply (91.3 % of the cases), while about one third of the cases have specific focus on water quality or water quantity. Moreover, improving the in-stream water supply, for instance to improve water flow regulation for hydropower production, is targeted in 53.3 % of all reviewed case studies (Martin-Ortega et al., 2012). A review of the IIED identified the maintenance of dry season flows, protection of water quality, and control of sedimentation as the main concerns of 61 studied payments for ecosystem services form forested watersheds (Landell-Mills and Porras, 2002).

During the negotiation process it is usually determined how to deal with conditionality and additionality in the application of the scheme. The outcome of this process has far reaching consequences on the eligibility of participants both on the provider and buyer side, on the outcome of the PHES in terms of land use changes and service provision as well as on the baseline assessment and monitoring of the program. Most of all, the latter aspects can increase both the transaction costs of the scheme and the opportunity costs of the service providers and will consequently lead to lower availability of financial resources for actual payments. The reduced availability of financial resources for actual payments and a possibly higher payment amount because of higher opportunity costs of providers will in total lead to a much smaller area to be contracted within the PHES scheme. Hence, higher stringency in conditionality can lead to a reduced volume of service flows obtainable for a given PHES budget (cf., Kroeger, 2012).

Besides the definition of conditionality terms, stakeholders involved in a PHES scheme need also to address the issue of additionality, thus, defining whether a certain measure can be qualified as an additional action as opposed to an action that the participant (i.e. provider) would have realized anyway without compensation. The additionality feature of measures in the context of PHES is the core of how the instrument relates to other policy instruments. However, this aspect will be discussed in the following chapter, based on an in-depth case study analysis.

Tognetti et al. (2006) stress that in considering additionality it should be kept in mind that the development of institutional capacity needed to effectively respond to watershed degradation can also have other benefits. According to Landell-Mills and Porras (2002), the institutional capacity building, that forms part of a PHES scheme can lead to social cooperation in other matters, development of skills, opportunities for clarification of land titles, and increase of scientific understanding, and environmental education (cf., Tognetti et al., 2006).

Hence, the negotiating process of a PHES scheme is basically about finding agreements in the context of conditionality, additionality and voluntariness of the instrument. This often implies an exchange of values and perceptions on what should and what not be compensated. Hence, PHES may be embedded in a policy mix among other regulatory instruments (e.g. establishment of protected areas based on command and control) such as those needed to deal with activities agreed to be ineligible for compensation. Furthermore, some actions may only be defined eligible for in-kind compensations, e.g. through the provision of material for fencing of restricted land use zones or capacity building to improve agricultural land use without continuous payments in the future.

The potential to engage private sector actors in conservation financing and activities is evidenced by a majority of private service buyers and seller, i.e. providers. The private sector, local and national government, local and international NGOs, community groups and donors are all participants in PHES schemes as buyers, sellers, intermediaries, brokers and providers of support services. Grieg-Gran and Porras (2012) report a generally good level of participation from smallholders and poor communities in local PHES schemes, because these schemes have been able to adapt to local circumstances, taking time to build up trust among the landowners and find ways around obstacles such as lack of clear land titles. For instance, in the PHES case of Los Negros in Bolivia, where the NGO Fundación Natura facilitated discussions between upstream and downstream landowners, followed by the introduction of a payment scheme with initial donor funding. The NGO used local recognition of landholdings where formal land titles were missing (Asquith and Vargas, 2007).

Tognetti et al. (2006) argue that the willingness of potential beneficiaries to pay for hydrological ecosystem services with public good characteristics (being difficult or expensive in limiting access to them), depends not only on the demand but also on beneficiaries' confidence in the effectiveness of the proposed management actions to ensure service provision. Moreover, potential buyer need to be sure that they will have access to the stream of benefits, i.e. access to the provided service. Hence, according to Tognetti et al. (2006), the value of watershed services will depend on:

- The integrity of ecosystem functions or processes that support service provision
- The scale at which impacts or benefits have economic significance and
- The effectiveness of institutional arrangements needed to insure provision and access.

Payment [ha/year]	Activities compensated	Hydrological services	Service buyer	Service seller	Location
US$ 45	Reforestation	Salinity control, freshwater supply	Downstream farmer association	Government and upstream landowners	Murray Darling Basin, Australia
US$ 230	Reduced-input farm management	Water quality control, freshwater supply	Perrier Vittel (Mineral water company)	Upstream farmers	Rhine-Meuse Basin, France
US$ 45-116	Protecting, sustainably managing and replanting forests	Freshwater supply, wildlife habitat, cultural heritage and identity	National Forest Office and National Fund for Forest Financing (FONAFIFO)	Private upstream landowners	Costa Rica
US$ 48	Protecting, sustainably managing and replanting forests	Hydropower, regulation of flows, sedimentation control	Hydropower company and FONAFIFO	Private upstream land owners	Sarapiqui watershed, Costa Rica
US$ 125	Soil conservation	Sedimentation / water quality control, regulation of flow	US Department of Agriculture	Farmers	United States
US$ 170	Watershed restoration	Freshwater supply, wildlife habitat	State of Parana	Municipalities and landowners	State of Parana, Brazil

Table 5.8: Examples of payment amounts for hydrological ecosystem services; from Smith et al. (2006) based on Perrot-Maître and Davis (2001); Kumar (2005)

Thus, within PHES schemes it may become necessary to address rivalry over access to a limited supply of hydrological services by developing of specific institutions, e.g. enforceable rules, through which access can be secured to those who are paying for its provision. These institutions may also define responsibilities for actions needed to insure that services are provided (cf., Ostrom, 1990).

However, besides the engagement of service buyers it is equally important to convince sufficient providers to join a scheme because hydrological ecosystems service provision is most likely to be characterized by threshold effects depending on the spatial and temporal extent of land use changes (see Section 4.2). Hence, Landell-Mills and Porras (2002) highlight that cooperation is a key requirement when it comes to the supply and demand for ecosystem services. A successful scheme, thus, may depend on strengthening cooperative and hierarchical arrangements allowing beneficiaries and providers to formulate group payment strategies and tackle free riding of non-paying beneficiaries.

Main recipients of payments for their role as service providers are landowners and farmers (mostly private, but in some cases public landowners or cooperatives of landowners). Martin-Ortega et al. (2012) identified this recipient group in 96.4 % of all cases in their review, however, it is often unclear whether farmers or land users also own the land since many case study documentations do not specify this[6]. In less than 5 % of the cases, the authors identified local and national NGOs as well as park administrations as sellers. The number of hydrological ecosystem service sellers varies widely. Martin-Ortega et al. (2012) document PHES schemes with only one seller to more than 800 sellers, with a median of 18 and an average number of 55 sellers.

The buyer side of PHES is generally more heterogeneous than the seller side. The largest buyer group are hydropower producers or domestic water users (Porras et al., 2008; Martin-Ortega et al., 2012). Irrigating agricultural producers and national or international NGOs are also common buyers but to a much lesser extent. Again the number of buyers differs largely, according to Martin-Ortega et al. (2012) ranging from one single buyer (e.g. a hydro power company) to 18,700 buyers (individual consumers in an association of domestic water users), with a median number of 8 buyers.

Examples of different negotiation results (payment amounts, activities to be compensated and hydrological ecosystem services aimed at) of PHES in practice are summarized in Table 5.8.

[6] Farmers are explicitly reported as service providers in 14.6 % of the cases, while landowners are mentioned in 77 % of the observations (Martin-Ortega et al., 2012).

The flexibility of PHES schemes with regard to the promotion of different actions to improve hydrological service provision and differentiation in respective payment amounts, i.e. compensations, is stressed by Martin-Ortega et al. (2012). 42 % of the cases reviewed by the authors include some kind of differentiation in actions to improve hydrological ecosystem services. Five different features were identified to classify for different compensation: (i) the type of activity (e.g. forest conservation, reforestation or improved agricultural practices); (ii) the land feature and type (e.g. slope or forest type); (iii) the number of practices applied; (iv) the surface covered; and (v) other features (e.g. type of ownership of the land, land use history or the status quality of the area) (Martin-Ortega et al., 2012). In most cases (75 %) the type of activity is the determining feature for the payments.

The second part of the instrument application incorporates the continuous realization of agreed actions, payments and monitoring of agreed actions. There are usually three mechanisms in place to achieve this: a supervising mechanism composed of a technical and a governance supervision, a financing mechanism and a payment mechanism. This typical PHES set up is illustrated in Figure 5.6.

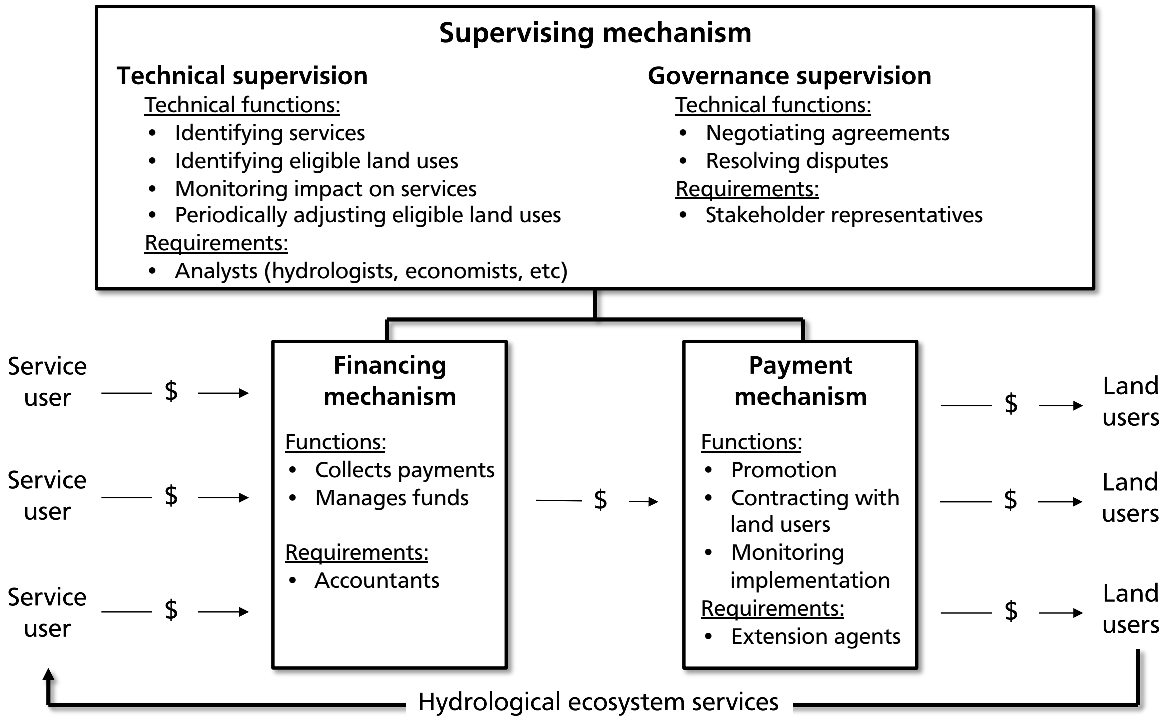

Figure 5.6: Organizational structure of a PHES scheme; based on Pagiola and Platais (2005)

The realization of agreed actions is often assisted by a technical supervision carried out by a specialized organization. For instance, capacity building on improved agricultural practices may be carried out by a governmental or non-governmental organization, such as a technical institute or a NGO experienced in agricultural land use improvements. This form of technical assistance may facilitate an upward connectivity to a higher governance level which provides assistance *on demand* of the parties involved in the PHES scheme. Assistance is often provided, although resources and capacities are scarce when a specified demand is placed. Moreover, technical assistance may also be provided in order to identify services and eligible land uses related to service provision, monitoring of impact on services and if necessary periodical adjustment of eligible land uses. At best technical experts from specialized organizations are involved in this supervision and potential service providers as well as buyers are consulted in order to find a consensus on land use linkages to service provision.

Besides the technical supervision, there is also a governance supervision needed to negotiate additional agreements on payments and to resolve disputes on ongoing agreements. The involvement of representatives of actual and potential service providers and buyers is crucial for successful governance supervision and durability of a scheme.

Furthermore, a financing and a payment mechanism are required. These mechanisms are not always organizationally separated. A financing mechanism is in place for the collection of payments and for managing the fund if necessary. The payment mechanism in turn has its function in contracting with the service sellers, monitoring the implementation of agreed actions and execution of payments. The latter may be done in a transparent manner to outsiders in order to promote engagement of potential service providers. The majority of PHES schemes studied by Martin-Ortega et al. (2012) involve periodical cash payments (mostly through specified PHES water fees) and includes different payment levels, depending mostly on the actions involved and the type of land use. In schemes where a large number of stakeholders (buyers and sellers) are involved, payments tend to be channeled through intermediaries (Landell-Mills and Porras, 2002). This allows buyers and sellers to contract out the negotiation and conclusion of deals, overseeing implementation and enforcing contracts. Moreover, intermediaries may be valuable for pooling funds from a group of beneficiaries and / or collecting user fees.

If agreed land uses do not lead to the desired outcomes of the service provision they may require revision. As a result of the conditionality criteria, the motivation for monitoring of land use and service provision arises. Thus, besides baseline information a PHES scheme, in an ideal case, also provides continuous information on land use and hydrological services. Monitoring implies additional transaction costs but is a prerequisite to prove the failure or success of introduced land use measures. Moreover, conditionality means that noncompliance has to be addressed and consequences, for instance exclusion from the payment scheme, have to be defined.

The monitoring of agreed actions of service providers in most cases is, as well the provision of ecosystem services, based on the compliance with agreed land uses (Engel et al., 2008). If a separate monitoring of hydrological services is lacking, it is impossible to verify that the promoted land uses result in the provisioning or maintenance of the ecosystem service that is being paid for (cf. Wunder, 2005). According to Grieg-Gran and Porras (2012), very few PHES schemes go beyond monitoring compliance with agreed land management practices to actually measuring trends in service provision. The meta-analysis of Brouwer et al. (2011) on 47 PHES schemes revealed that for 47 % of the schemes a monitoring on quantified objectives was documented. In this context, important issues have to be dealt with in order to find agreements between providers and beneficiaries in terms of additionality and conditionality of payments.

Pagiola and Platais (2007) stress the particular strength of local PHES schemes by directly involving local actors as the actors with the most information about the value of the service. Thus, local actors have a clear incentive to ensure that the mechanism is functioning well, can observe directly whether the service is being delivered, and have the ability to re-negotiate or even terminate the agreement if needed (Engel et al., 2008). Local actor involvement, hence, makes efficiency improvements more likely by reducing transaction cost (Pagiola and Platais, 2007).

5.2.2 Actors and institutional roles in the governance and management of PHES schemes

The importance of actor-oriented incentives to overcome problems of fit and interplay in order to operationalize IWRM from bottom-up has been highlighted in Section 3.4. Thus, based on the identification of typical PHES implementation processes, this section focuses specifically on actors and institutional roles in the governance and management of PHES schemes.

According to Smith et al. (2013), there are typically four principal stakeholder groups involved in a PHES scheme:

- **Buyers and beneficiaries** of ecosystem services who are willing to pay for them to be safeguarded, enhanced or restored.
- **Sellers and providers**; land and resource users whose actions can potentially secure supply of the beneficial service
- **Intermediaries and facilitators**; who can serve as agents linking buyers and sellers and can help with scheme design and implementation.
- **Knowledge providers**; these include resource management experts, valuation specialists, land use planners, regulators and business and legal advisers who can provide knowledge essential to scheme development.

In the local schemes that are considered here in the context of IWRM operationalization, the buyer side always includes actual users of the provided hydrological ecosystem services, but may be complemented by national governments that buy services as part of a national scheme. Typical service buyers are local public or private water suppliers, municipalities, hydro power producers, industry and irrigation agriculture. Thermal power plants, which require large quantities of water for cooling, have so far not been engaged as service buyers. The service buyers generally join the PHES scheme voluntarily, but water end users (e.g. domestic water users) are often forced to buy the service, i.e. to pay a fee, through their water suppliers. However, the service providers or sellers, in most cases private farmers or forest owners, communal landowners, and in some cases government or NGOs managing protected areas, do generally join a scheme on a voluntary basis (Porras et al., 2008; Brouwer et al., 2011). Blackman and Woodward (2009) analyzed the drivers for participation in a PHES scheme of the private hydroelectric sector in Costa Rica based on semi-structured and open-ended interviews with participants and non-participants. They revealed that, both participants and nonparticipants, besides stating that the main benefit of participation was forest protection and the provision of environmental services, also emphasized improved relations with local communities and with government regulators.

Table 5.9 summarizes the main stakeholders from different sectors and highlights their principal incentives for participating in a PHES scheme. While private sector stakeholders perceive incentives in form of reduced costs and increased security of service provision, the public sector and local communities see additional incentives in improvements, when complying with their role, and in gaining more influence in decision-making processes.

Sector	Stakeholder	Incentives for participation
Public	Water companies, local governments	Water quality, water regulation, avoided sediment costs
	Hydropower generation companies	Water flow regulation, avoided sediment costs
	National environmental authority	Strengthening, financing and fulfilling protected area management plans, resource conservation
	Local environmental authorities	Strengthening, financing and fulfilling protected area management plans, resource conservation
	Water authorities	Management of watersheds, resource conservation
	Irrigation zones	Water regulation, avoided sediment costs
Private	Water companies	Water quality, water quantity, water regulation
	Hydro power generation companies	Water regulation, avoided sediment costs
	Bottled water and soft-drink companies	Water quality, water regulation, avoided sediment costs
	Agricultural associations	Water regulation, avoided sediment costs
	Industries	Water regulation and water quality
Local Communities	River associations, water boards, irrigation boards	Participation and investment decision-making, resource conservation
	Indigenous communities	Participation and investment decision-making

Table 5.9: Stakeholders participating in Payments for Hydrological Ecosystem Services schemes and respective incentives for participation; based on Calvache et al. (2012)

Greiber (2009) identified potential stakeholder groups involved in the planning, negotiation and implementation processes of PHES. These stakeholders as well as their typical roles are described in Table 5.10.

Figure 5.7 illustrates schematically the level of influence and interest of different stakeholders participating in an PHES scheme based on findings of Calvache et al. (2012) in the context of water funds. While some important intermediaries such as multilateral cooperation agencies and international NGOs have a high level of interest in PHES, others usually have a stronger level of influence, for instance public authorities at different governance levels. Hence, it is not surprising that these actor groups act mutually to implement PHES schemes. However, service beneficiaries as well as local NGOs, protected area agencies and community-based organizations often combine a high level of influence and interest in PHES, the former as potential buyers and the latter as important intermediaries.

The PHES review of Martin-Ortega et al. (2012), revealed the presence of intermediaries in a large majority of schemes (almost 80 %) with almost a quarter of schemes where various intermediaries are involved. According to the authors, most often local NGOs (about 25 %), followed by multi-stakeholder trust funds (13 %; e.g. FONAG in Ecuador and the FIDECOAGUA in Mexico) and municipalities (10.5 %) are involved as intermediaries.

Intermediaries and facilitators often play an important role in the initiation, the implementation process and the subsequent execution of PHES schemes. According to Greiber (2009) and Hedden-Dunkhorst et al. (2010), they may comprise the following central technical and coordinating tasks:

- Scientific advice to project developers, particularly regarding the identification of expected downstream services
- Design of payment mechanisms, feasibility studies, management plans and monitoring systems
- Facilitation of negotiations among all stakeholders by linking potential buyers and sellers while involving other relevant stakeholders in the design process
- Stimulate and institutionalize inter-sectoral coordination and cooperation (e.g. between water, forestry, social development, economic sectors)
- Provide information and juridical support to contract parties and stakeholders (e.g. on environmental monitoring or the negotiation of legal transactions)
- Promote transparency and local acceptance for the scheme beyond contract parties
- Manage expectations and resolve occurring conflicts through stakeholder dialogues
- Land management capacity-building
- Collection of hydrological data and
- Contract administration, allocation of funds and payments.

Most PHES schemes are initiated by NGOs or international agencies, sometimes with municipalities as their counterparts. The initial bio-physical assessment and socio-economic valuation are often carried out by these actors as well or through

Stakeholder group	Description, typical examples and roles
Beneficiaries	Private or public instances who have a demand for the provision of hydrological ecosystem services
Providers	• Private landowners; individual owners with clear and undisputed property rights; • Communal landholders; farmers living on or drawing their livelihood from communal property; • Private reserves; whether an individual or group, private entities registered as reserves and committed to ecosystem conservation are the third most common supplier of watershed services; • Governments or non-governmental organizations (NGOs); land owned and managed by governments or NGOs for conservation purposes; • Informal occupiers of public lands; farmers living on public property, oftentimes designated as a protected area, who may have long-standing rights to the land.
Donors	• Government; providing municipal and national government funding; • Private sector; making voluntary and required payments for water-related ecosystem services; • Private individuals; paying household and agricultural fees for water use; • Charitable foundations; making donations from their assets. Beneficiaries and donors will often overlap.
Intermediaries	Intermediaries (governmental entities, international agencies or NGOs) may link donors, beneficiaries and suppliers of water-related ecosystem services, and aid in the development, administration or operation of a PHES scheme.

Table 5.10: Potential stakeholders involved in the planning, negotiation and implementation processes of Payments for Hydrological Ecosystem Services; based on Greiber (2009)

universities or research institutes. Intermediaries and facilitators, including national or local government agencies, environmental NGOs, development NGOs and funding institutions, also operate at various stages of the process from initial stakeholder dialogue to design, implementation and operation of the PHES, and are involved in most PHES. Porras et al. (2008) identified in their global review on PHES schemes only "few schemes where the suppliers and the buyers make arrangements for payment without the help of a facilitator at some stage of the process". Some intermediaries act primarily as facilitators having a transitory character, for example in assisting during initial stages of a scheme through facilitating dialogue or information but withdrawing when the scheme is established and others take ownership (Porras et al., 2008). International development agencies and NGOs often take this transitory role.

Porras et al. (2008) in their PHES review stress not having identified any facilitator operating as a market player, taking on risk by buying hydrological services and selling them on to different users. This again documents the limited market-like, i.e. purely Coasean nature of PHES schemes. However, intermediaries often play an important role in determining payments through negotiation at the local level. According to Porras et al. (2008), here they may be needed to create a negotiation forum and to assist the weaker party, usually the service providers, with a negotiating strategy. In many schemes in South and Central America trusts have been created as intermediary structures.

Moreover, intermediaries are regarded as important both in the design and the operation of PHES (Porras et al., 2008). The diversity of intermediaries and stakeholders involved with different tasks often leads to a variety of partnerships between different types of organization, governmental and non-governmental groups, academia and international interest groups. Buric et al. (2011) in the case of PHES schemes for urban beneficiaries characterize them as "multi-stakeholder affairs involving national and local (regional / municipal) governments, community groups, individual landholders, commercial enterprises, non-governmental organizations and various donors". While governmental entities play an important role in providing information and capacity, local intermediaries (e.g. NGOs, user associations or municipalities) are of particular importance to strengthen the link between providers and users at the local level in a flexible manner (Porras et al., 2008). Moreover, Porras et al. (2011) argue that the ability to find an intermediary to group small providers, often dispersed, and thereby keeping transaction costs low is often necessary to safeguard the development of socially equitable PHES.

The role of facilitators and intermediaries, especially public authorities, is also very important in the context of monitoring. They can support monitoring and improved service provision measures, e.g. through capacity building for sustainable agricultural practices, with the result of leaving more of a PHES budget for actual payments and reducing the participation cost.

Especially in Central America, local municipalities play a special role as service buyers, facilitators and intermediaries. Besides their role as service buyers in case of municipal water supply, they also play an active role in facilitating information on land and water uses / users and often provide the legal basis for a scheme through municipal decrees (Pérez et al.,

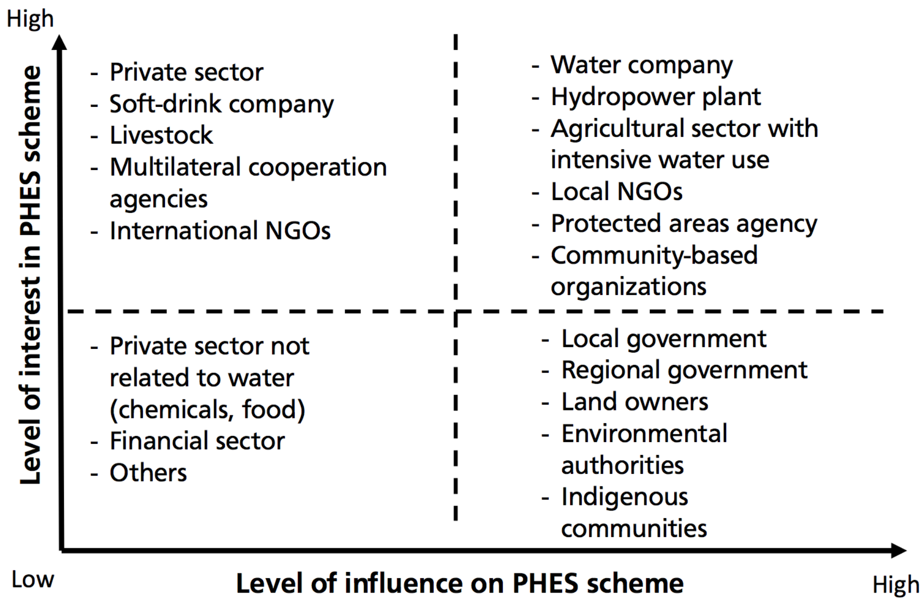

Figure 5.7: Outline of stakeholder influence on and interest in a PHES scheme; based on Calvache et al. (2012)

2002; Mejías Esquivel and Segura Bonilla, 2002). Moreover, municipal involvement may be needed to resolve disputed or informal land uses and rights.

The Swiss Agency for Development Cooperation (SCD), one of the most important donors of local PHES with its Programa para la Agricultura Sostenible en Laderas de América Central (PASOLAC) initiative in Central America, proposes the establishment of Ecosystem Service Commissions (Comisión de Servicios Ambientales; COSA) for the management of payment schemes in form of a custodian for the payment fund. According to the SCD, this inter-sectoral and inter-institutional commission should be composed of representatives of all stakeholders involved in the PHES scheme. For instance, in the context of a PHES aiming to protect drinking water resources from soil erosion and resulting sedimentation, the commission would be composed of upstream land users, representatives of the municipality, the water supply company, the civil society, a technical expert and a representative of the organization in charge of the payment fund. The commission is in charge of assigning responsibilities for managing the payment fund, supervision of payment agreements and execution as well as coordination of agreed activities for the implementation of the scheme (Pérez et al., 2002).

Water funds in South America have a similar intermediary structure in the form of fund boards. According to Goldman-Benner et al. (2012), fund boards are composed of diverse stakeholders, including water users and service providers, and makes decisions about how to use funds. In bringing together different stakeholders with multiple objectives, joint decision making may divert service buyers' investments from the services they most care about. Hence, Goldman-Benner et al. (2012) stress that this may lead to large-scale, integrated water management with partnerships that "provide a way to include social criteria as well as biophysical and economic measures in defining optimal investment portfolios".

5.2.3 Constraints for the development of Payment for Hydrological Ecosystem Services

Just as any other policy instruments, PHES schemes have instrument specific weaknesses and require certain circumstances in order to be successful. Criticism concerning the threat of commodification of nature through the introduction of ecosystem services into a classical market economy have already been addressed in Chapter 4 and this criticism applies equally to PES as expressed with the objection of the PES instrument when purely market-based for instance by McCauley (2006); Robertson (2006); Redford and Adams (2009); Kosoy and Corbera (2010).

Redford and Adams (2009), for instance, outline seven problems with ecosystem services which they consider necessary to be addressed in order to make the role of payment for ecosystems services in conservation clearer and arguments for conservation stronger:

(1) The real risk that economic arguments about services valued by humans will overwrite and outweigh non- economic justifications for conservation.
(2) Erroneous assumption that ecosystem services are necessarily benign implying a danger to limit the focus on regulating ecosystem services to times and in flows that match human needs.

(3) Environmental policy based on the optimization of ecosystem-service values will not necessarily lead to the conservation of biodiversity, e.g. preference of exotic instead of native species for the provision of services.
(4) Maximization of single-service provision leading to increased ecological brittleness.
(5) Markets only exist for a certain range of ecosystem services, and some services are not amenable to pricing or valuation, e.g. where a valuable service is provided by a biodiverse ecosystem (e.g., water yield from a forested catchment), where that ecosystem is close to a major consumer, and where institutions exist to enable those consumers to pay for the service they receive, ecosystem services may provide a powerful stimulus for conservation. Elsewhere, they will not.
(6) As ecosystem services become increasingly scarce and valuable, people will compete to gain control over flows of services and the ecosystems that provide them.
(7) Impacts of climate change on ecosystem service delivery are unknown and may cause identified cause-effect relationships to become obsolete.

Furthermore, Amezaga (2006) highlights typical constraints for the implementation of benefit-sharing mechanisms which may also be true for PHES. According to Amezaga, these constraints range from the need to find a compromise among conflicting interests over the distribution of cost and benefits to institutional challenges and up-front costs of engaging stakeholders in initial planning stages. In order to be successful, a common understanding and agreement about the nature of the expected impact, the approximate magnitude of cost and benefits, and the areas of uncertainty must be clear for all stakeholders.

Besides the general criticism on PES as a policy instrument, there are some principal constraints to the development of PHES schemes. Landell-Mills and Porras (2002) identified transaction costs, demand-side constraints and supply-side constraints as particular constraints toward PES development.

Transaction costs have considerable influence on the development of a PHES scheme since these costs reduce the budget available for payments to service providers. Compared to other ecosystem services, e.g. carbon sequestration, the cost of establishing a payment scheme for hydrological ecosystem service can be high due to the specific cost of identifying potential providers and buyers, negotiating to implement a scheme, monitoring and analyzing service delivery, documentation and record keeping and administration of contracts (Landell-Mills and Porras, 2002). This is a result of the nature of hydrological ecosystem services, the possibly large number of participants involved and often the absence of cooperative structures to build negotiations on. Landell-Mills and Porras (2002) regard multiple-stakeholder transactions, lack of cost-effective intermediaries and poorly defined property rights as main reasons for the particularly high transaction cost in the context of PHES schemes.

PHES schemes between upstream and downstream communities frequently require a large number of participants, especially on the side of service providers in order to become feasible. Hydrological threshold effects imply the coverage of a minimum area of the river basin, often involving large numbers of land users, to cause the desired effects on service provision. Broad participation is also essential to avoid free-riding in consumption and to convince beneficiaries to pay (Landell-Mills and Porras, 2002). The number of potential participants as well as related transaction costs usually increases with the size of the river basin. Several authors, Ostrom (1990); Rhoades (1998); Magrath et al. (1997) among others, have highlighted the cost of multi-stakeholder participation. Besides transaction costs originating from multi-stakeholder participation, additional costs arise from building organizational structures of intermediaries who bring buyers and seller together. These intermediaries often require strong managerial, financial and technical skills. At best it can be built on existing organizations, where these are absent often NGOs or community organizations are strengthened in their capacity to function accordingly. However, this often needs higher investments (cf., Landell-Mills and Porras, 2002).

Another factor influencing transaction costs are property rights. According to Landell-Mills and Porras (2002), in most cases insecure tenure remains a principle constraint against the development of schemes. However, in their review the authors point out a number of innovative approaches to define property rights, e.g. in the form of transpiration credits, salinity credits, watershed management contracts and conservation easements.

Examples for the transaction cost of two PHES schemes in Madagascar are summarized in Table 5.11.

PHES scheme	**Area** [ha]	**No. of beneficiaries**	**No. of providers**	**Transaction costs**
Drinking water supply of the city of Fianarantsoa	3,500	150,000	196	€ 15,456 *Compensation: € 77,037*
Drinking water supply of the city of Andapa	910	30,000	32	€ 16,419 *Compensation: € 151,481*

Table 5.11: Parameters and transaction costs of PHES in Madagascar; based on (Bidaud et al., 2013)

Several factors that undermine demand for hydrological ecosystem services are mentioned by Landell-Mills and Porras (2002) as further constraints to scheme development. Of course, in order to be willing to pay for the provision of hydrological ecosystems services, potential buyers need to be aware of potential benefits from specified land uses. Lack of confidence in or scientific evidence of cause-effect relationships between land use and service provision may significantly lower the willingness to join a payment scheme on the buyer, i.e. demand side. Moreover, Landell-Mills and Porras (2002) stress that "where beneficiaries are not involved in designing the payment system and ensuring against free-riding, they may be unsupportive". Participation of potential service providers is equally important as payments should meet their needs to engage them as well. Thus, the active participation of these key stakeholder groups in the design and implementation of PHES schemes is essential. This has been explicitly acknowledged by the US Environmental Protection Agency, for instance, by issuing specific guidelines on stakeholder participation in its Water Quality Trading Assessment Handbook (EPA, 2004). Furthermore, Landell-Mills and Porras (2002) highlight the lack of willingness to pay as an additional constraint to the development of a PHES scheme because of undermined demand and stress two mayor reasons for this from literature:

(1) Resistance of stakeholders that are used to receiving watershed protection services for free. This is particularly damaging where more powerful entities are determined to face down efforts to force them to pay the full costs of water provision.
(2) Lack of finance, especially where the government is the buyer (in Vietnam, for instance, government payments for watershed protection are too low to attract the necessary landowner participation; (cf. Sikor, 2000)).

Transaction costs and factors that undermine demand are not mutually exclusive. Often increasing transaction costs result in undermined demand, for instance.

Finally, Landell-Mills and Porras (2002) also regard factors that undermine supply as potentially hindering PHES scheme development. The authors acknowledge a widespread lack of awareness of potential opportunities to exploit positive externalities. Even when land users are aware of their role in providing services, downstream beneficiaries are most likely willing to pay when they perceive their water supplies as threatened. However, Landell-Mills and Porras (2002) see potential for land users to be more proactive in bargaining for payments by taking positive externalities into account when making land use decisions. This allows them to determine the minimum payment they are willing to accept in order to abandon their plans. With this information they are in a position to initiate a bargaining process. A final aspect that may undermine supply is a lack of credibility in service delivery. It has to be clear how service providers will alter service delivery. If the provider side wants to establish a payment scheme, it may have to offer some form of insurance scheme to minimize risks to downstream buyers where reliable, site-specific hydrological data illustrating clear land use-service provision linkages is missing.

Apart from general criticism on PES (cf., Redford and Adams, 2009) and constraints to scheme development (cf., Landell-Mills and Porras, 2002), there are also critical reviews specific to the functioning and implementation requirements of PES. Wunder et al. (2008), for instance, examined 14 PES case studies to identify whether the scheme, i.e. program, succeeds in generating the desired ES. The authors identified four critical issues related to the function of PES schemes and several other issues with regard to achieving environmental goals. The four issues related to the functioning of schemes are: enrollment of potential ecosystem service providers, compliance of ecosystem service provision and additionality as well as land use-ecosystem service provision linkage.

In general, enrollment of potential hydrological service providers is not difficult in practice as long as there is no significant mistrust concerning buyer or intermediary compliance. According to Wunder et al. (2008), in most cases applications from service providers exceeded by far the available funding. However, high disposition of potential service providers is not always given for the most critical provisioning areas if opportunity costs significantly exceed the payments offered.

The issue of compliance with agreed actions is an important one and requires some kind of monitoring. The common proof of compliance is achieved through site inspections, in the case of the FIDEICOMISO scheme in Mexico additionally through remote-sensing satellite imagery since it is integrated in the national PHES scheme. Monitoring implies transaction cost which can vary widely depending on the characteristics (e.g. size and accessibility) of the area to be monitored as well as on the temporal interval. It is important that service providers and buyers agree on a feasible practice of monitoring. Moreover, agreements on monitoring require agreements on sanctions in order to ensure compliance. The principal sanction in most PHES schemes, according to Wunder et al. (2008), is the loss of future payments, either temporarily or permanently, and in some cases reimbursement of previous payments. However, there are no systematic studies on the effectiveness of different types of sanctions to achieve compliance.

With regard to ensuring service provision compared to a situation without a payment scheme, compliance is strongly related to additionality because it considers whether agreed actions (or non-actions) would have been done even in the absence of payments. In practice, situations without intervention in form of business as usual scenarios are extremely difficult to determine. Wunder et al. (2008) document just one case where additionally is formally quantified for various ex ante scenarios. However, more attempts to assess additionality have been realized, most of them in Costa Rica, in an ex post manner with divergent results (Ferraro and Pattanayak, 2006; Sánchez-Azofeifa et al., 2007). Experience of many user-financed PHES schemes provides reasonable good evidence of high additionality. (Wunder et al., 2008) report that in the case of Pimampiro in Ecuador, for instance, previous deforestation trends were reversed in the scheme area, while deforestation continued apace in surrounding areas. Additionality can most easily be achieved, i.e. monitored where explicit land-use changes are agreed, for instance reforestation or structural changes in agricultural practices. Land use changes which are markedly different from observed land use trend may also have higher additionality. In some cases additionality can be assumed if land use changes occur within the scheme area, while promotion of the same land use changes are not achieved elsewhere, despite being promoted as well. Wunder et al. (2008) provide such an example with the success of the PROFAFOR PES in establishing more than 22,000 ha of forest plantations on degraded lands in Ecuador, while a variety of traditional subsidy-based reforestation programs elsewhere in the country failed to achieve significant results.

Improvement of the provision of hydrological ecosystem service is still not secured based on provider enrollment, compliance and additionality alone. Additionally, the assumed cause and effect relationship between land use changes and service provision has to be true as well. Reis et al. (2007) stress that this is the most important biophysical aspect influencing the success of a PHES scheme. As has been discussed in Section 4.2, clarifying these cause and effect relationships is a complex issue for hydrological ecosystem services, especially when the relationship has been little studied. There is much controversy whether assumed cause-effect relationships eventually are true, especially when considering the role of forest for quantitative aspects of water provision (Chomitz et al., 1999; Bruijnzeel, 2004; Calder, 2004). There are just a few PHES schemes where a thorough scientific investigation of land use linkages to hydrological service provision has been realized. Exceptions are South Africa where the water consumption of invasive alien species has been well-documented (cf., Turpie et al., 2007) or the Los Negros PHES in Bolivia where the role of cloud forest in increasing dry season flows was studied(cf., Le Tellier et al., 2009). However, although land use linkages to service provision are not always clearly identified it is important that providers and beneficiaries agree on some kind of principle assumptions of them which have to be verified in the future through measurements and monitoring. Moreover, in areas deemed to be suitable for conservation in order to secure continued service provision, the precautionary principle provides sufficient reason to do so. This precautionary logic has been the basis for many water user-financed PHES in Costa Rica (cf., Wunder et al., 2008). Some relationships between land use and service provision, however, are reasonably well established, e.g. erosion control by perennial vegetation (Hope et al., 2007). As most PHES schemes lack a profound scientific basis of cause-effect relationships it is very much advisable to provide for sufficient monitoring and measurements in order to learn more about them and to react accordingly.

This form of continuous learning has been repeatedly stressed by *adaptive* approaches of IWRM operationalization (see Section 2.4). Wunder et al. (2008) argue that it seems more reasonable to expect user-financed schemes to adapt this learning approach with a focus on monitoring because users have a strong incentive to ensure that their money is spent effectively. Moreover, the smaller scale of user-financed PHES makes it easier to observe whether the desired services are being generated or not. Hence, PHES schemes provide an incentive to gather site-specific hydrological data. In the long run, more robust explanations for land use and service provision linkages will be either beneficial on the willingness to pay or lead to a revision of previously assumed linkages in order to improve the scheme's outcome. In the end, increasing knowledge on land use linkages can even lead to discarding a PHES scheme if service provision may not be sufficiently affected by land use changes alone. Hence, the instrument itself may control its suitability with regard to the problem to be solved.

Besides these four issues determining the provision of desired hydrological ecosystem services, Wunder et al. (2008) consider three additional aspects that help to identify (i) whether ecosystem services are provided on a long-term basis (*permanence*); (ii) whether environmentally-damaging land uses are displaced to areas outside the scheme area (*leakage*); and (iii) whether the program creates *perverse incentives*. Duke et al. (2011) also regard leakage and perverse incentives as well as additionality as particular pitfalls of PES schemes.

Permanence of a PHES scheme depends on its continued and sustainable financing. For user-financed schemes continuous payments depend on maintaining buyers' willingness to pay. As long as buyers are satisfied with service provision or the agreed land use changes, they will probably continue to finance a scheme. It is important, thus, to improve the available knowledge on service provision and the impact introduced land use changes have on it (Pagiola and Platais, 2007). The permanence of schemes supported by governmental financing again depends on a continued governmental budget allocation for this purpose. During the execution of a PHES scheme, permanence can also be compromised when participants exit the scheme, e.g. by not renewing their contracts or by violating agreed terms. Incentives to exit a scheme on the provider side can be present when potential benefits of alternative activities increase. Theoretically, payments may be adjusted to increasing or declining opportunity costs of service providers, however, most PHES schemes have been in operation for too short a period of time to confront this situation in practice (Wunder et al., 2008). Some authors, for instance Swart (2003), have doubts whether actions towards service provision will be maintained once payments end. A feasible way to increase the probability of permanence is presented by the promotion of services providing land uses which also provide permanent on-site benefits, e.g. schemes focusing on planting trees base expectations of permanence beyond the end of payments on the expected benefits from sustainable timber harvest. According to Wunder et al. (2008), some schemes apply "short-term payments on the premise that the practices being supported are privately profitable once established, and thus will be retained". This form of incentive can be suitable to introduce transitions toward more sustainable land use.

Wunder et al. (2008) further mention *leakage*, a situation where environmentally-damaging activities are merely displaced rather than reduced, as another concern regarding the effectiveness of PES. This problem is especially important for global services like carbon sequestration. However, for spatially explicit services such as hydrological ecosystem services, the presence of leakage will depend on the specific scale of intervention. If undesired land use is displaced outside the considered river basin, than leakage is at least not affecting service provision but may have negative impacts elsewhere. However, case study documentations of PHES schemes have not yet raised concerns about leakage. Usually monitoring provides a good means to reduce the risk of leakage and particular contract design addressing this issue can further reduce this risk. The relatively small size of user-financed PHES schemes makes them leakage unlikely (Wunder et al., 2008).

Furthermore, PHES schemes have to take care not to create *perverse incentives* to potential service providers. A typical example of such an undesired incentive is when offering payments for reforestation could induce deforestation (Engel et al., 2008). Pagiola and Platais (2007) stress that schemes focusing on additionality are particularly at risk of creating perverse incentives if payments are offered only in the presence of clear threats of degradation, then potential applicants may be induced to create such threats. The Clean Development Mechanism (CDM) provides an example how to avoid this kind of incentives through careful contract design in specifying that only areas deforested prior to 1990 would be eligible to sell carbon credits from reforestation. However, PHES schemes can also introduce additional positive incentives, for instance if deforestation is declared an exclusion criteria for receiving payments in the future, hence, even non-participants may retain forests in order to maintain the option to participate. Tattenbach et al. (2006) highlight that a PES scheme in Costa Rica has this option value creating effect for forests. To avoid the problems of leakage and the creation of perverse incentives, Duke et al. (2011) argue that PES schemes require strong institutional capacity facilitating monitoring and oversight.

The PES instrument has also raised concerns about its influence on equity, especially on the poor in developing countries. These concerns are often expressed in the context of Coasean PES with focus on efficiency. Specific concerns relate to the ability of the poor to participate in schemes. One important obstacle to participation of the poor is the transaction cost of participation. Therefore, Costa Rica established a system of collective contracting to reduce participation cost for poor participants. Insecure land tenure can also impede the participation of the poor since they may only have use rights to the land without ownership. However, in local PHES schemes the problem of land tenure has been successfully overcome. Moreover, Zilberman et al. (2008) stress the importance of land distribution and service productivity as important criteria where the poor land user may benefit. Investment costs are another issue which may cause problems for poor participants. Specific land use changes, e.g. reforestation, may require initial investments which poor participants may not be able to cover. This problem is often addressed by upfront in-kind payments of the material required to comply with the agreed land use.

Pagiola et al. (2005) argue that PES may address poverty mainly by making payments to poor natural resource managers in upper watersheds. However, the impact depends on how many PES participants are in fact poor, on the poor's ability to participate as well as on the amounts paid. The authors stress that here can be important synergies with poverty reduction goals when the scheme's design is well thought out and local conditions are favorable, while there may also be adverse effects where property rights are insecure or if less labor-intensive practices are encouraged. Moreover, Pagiola et al. (2005) point out that designing payment mechanism so as not to exclude poor land users is probably the most important step in reducing adverse effects on the poor. Keeping the transaction costs as low as possible and developing creative responses to problems of insecure tenure can achieve this. Where strong local organizations, e.g. community groups or NGOs, assist in organizing participants and providing a forum for discussing solutions to problems as they arise, pro-poor scheme design is easier (cf., Pagiola et al., 2005).

In any case, it should be kept in mind that PHES schemes are not primarily designed for poverty reduction but to address environmental management problems. Hence, benefiting the poor is only a possible beneficial side-effect. It cannot be a guiding principle for provider selection since areas of potential service provision are not necessarily correlated with poor lands user or owners. However, beneficial impacts of PHES on poor participants have been documented by Suyanto et al. (2007) for a case in Indonesia where service providing land uses, here agroforestry systems of shaded coffee plantations, also created on-site benefits to service providers. Bond and Mayers (2010) claim that "there is little evidence of PWS doing any harm to poor people" based on their multi-country evaluation of case studies. However, payments are usually closer to the WTA of providers than to the WTP of buyers since the latter is often more difficult to identify. Hence, it is assumed that payments, in most cases, do not significantly improve household incomes of service providers but schemes may offer new opportunities to gain income which in the long-term can be significant. Landell-Mills and Porras (2002) summarize what they consider the most important constraints to the development of pro-poor PHES schemes (see Table 5.12).

Constraint	Explanation
Costs of organizing multiple-stakeholder schemes	Poor people tend to hold smaller plots, making co-ordination of supply more complex and costly. Poor beneficiaries may be more numerous and water use be informal and unregulated, making it more costly to incorporate them into payment schemes
Poorly defined / insecure property rights	Poorer groups tend to be the worst effected by insecure tenure as they lack the contacts, power and know-how to formalize their property claims
Lack of stakeholder participation	In negotiations around instrument design and payments, poorer individuals and groups are often most vulnerable to exclusion because of lack of skills to ensure their voice is heard and lack of political representation
Ability to pay	Where potential poor beneficiaries lack financial resources to pay, they have no influence over the allocation of resources to watershed protection
Low awareness of service payment opportunities	Poor people tend to be least well-educated about payment opportunities for hydrological services, and least able to initiate bargaining with downstream beneficiaries. They are less powerful and lack essential marketing, negotiation and coordination skills
Lack of credibility in service delivery	Where landowners' property rights are insecure, they are in a weak position to promise delivery of services. It is more difficult for poor service providers to guarantee forest protection since they need to maintain flexibility to respond to unexpected shocks

Table 5.12: Constraints to the development of pro-poor Payments for Hydrological Ecosystem Services schemes; based on Landell-Mills and Porras (2002)

Embedding PHES schemes with a policy mix of other governmental regulations is important to prevent that PHES schemes reinforce entrenched inequities, and to promote participation opportunities to weaker groups (Landell-Mills and Porras, 2002).

5.3 Addressing the institutional challenges of operationalization and provision of incentives toward IWRM

The experiences with local PHES schemes revealed several instrument characteristics which are promising in the context of IWRM operationalization. Three aspects are particularly important: the incentive provided by PHES schemes to different actors to overcome problems of fit and interplay (see Sections 3.1 and 3.2), the way how the instrument addresses operational constraints (Section 3.3), and how the instrument fits into existing policy mixes (Section 3.4).

With regard to the provision of incentives to solve problems of fit and interplay in the IWRM context, PHES bear several specific advantages compared to other policy instruments. If conditionality and additionality are part of PHES design, the typical implementation process as described in Section 5.2.1 provides several important incentives in the context of IWRM operationalization:

(1) Realization of bio-physical assessments and monitoring in order to understand cause-effect relationships between different land uses and hydrological service provision.
(2) Involvement, interactions and agreements between multiple stakeholders (ecosystem service providers and buyers as well as additional intermediaries) across sectors and administrative boundaries.
(3) Long-term commitments for service provisioning and financing as well as the establishment of organizational structures.

The PHES implementation process in the context of IWRM operationalization is illustrated in Figure 5.8.

The realization of bio-physical assessments and the involvement, interactions, and agreements between multiple stakeholders basically represent the ecosystem service approach to solve the governance problems of institutional fit and interplay in IWRM implementation, as discussed in Section 4.4. The outcomes of this process are, in an ideal case, spatial land use planning based on a river basin or a convenient part of it as planning unit, an established stakeholder dialog, a baseline assessment and a monitoring program of hydrological ecosystem services as well as organizational structures for a context-specific management. As the documentation of typical PHES implementation revealed these outcomes are not always achieved due to design limitations of the scheme or a lack of resources. However, although bio-physical assessments and economic valuation could be improved, the stakeholder dialog is actually most important as it facilitates negotiations on land uses, even though it is based on commonly shared beliefs rather than scientific evidence. For the permanence of PHES schemes, however, it is important to monitor land use and / or service provision in some mutually agreed form. This allows for continuous learning on bio-physical cause-effect relationships.

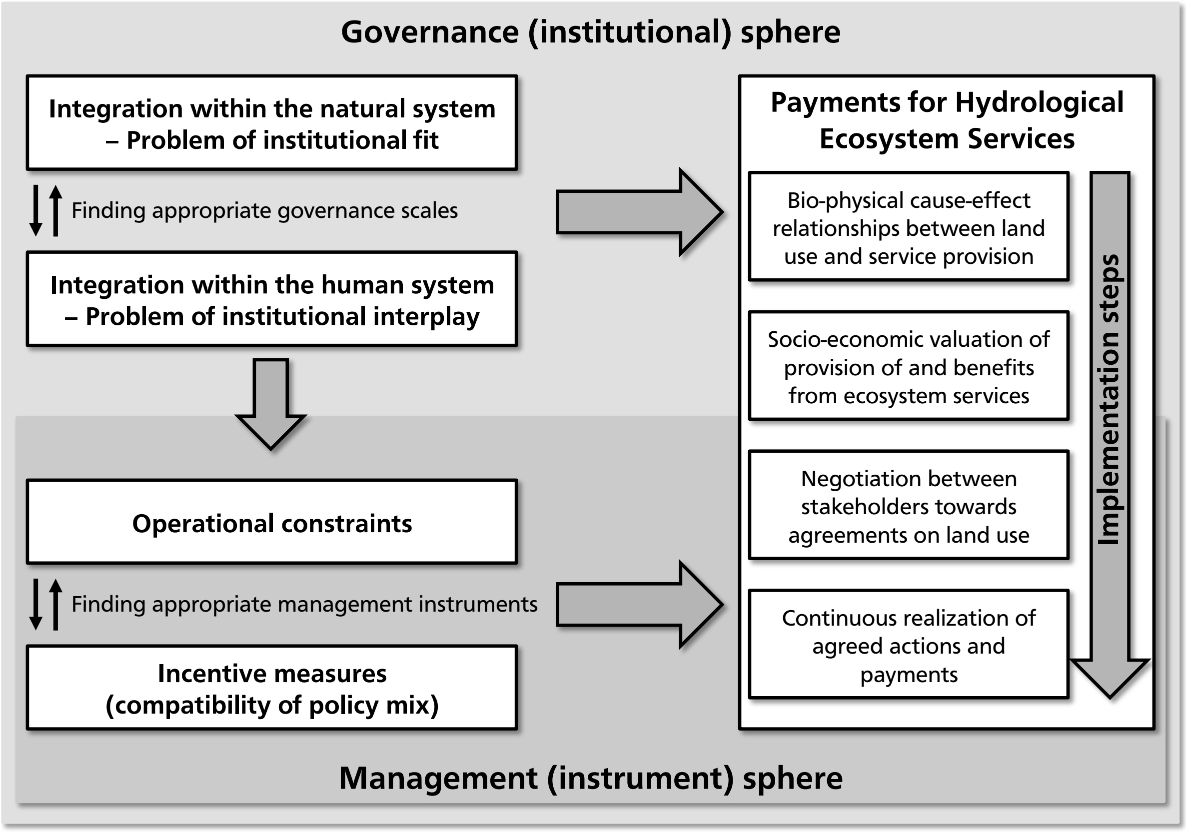

Figure 5.8: PHES implementation in the context of IWRM operationalization (Author's work)

It has to be stressed that the stakeholder negotiation process and the agreements on land uses occur within practical problem contexts which means that the outcomes are both context-specific and problem-oriented. Hence, they represent an *expedient* form of resource management (cf., Section 2.4) from bottom-up which works best when all stakeholders are involved. Stanton et al. (2010), for instance, highlight the role of PHES as a mechanism for collaborative action because the implementation process entails policy decisions requiring negotiations between multiple stakeholders including providers, beneficiaries, and intermediaries. According to Stanton et al. (2010), it is the negotiating process among the different actors that often leads to more effective and sustainable agreements which are vital to better land use and planning. Moreover, PHES implementation facilitates more awareness among all actors involved of the inherent benefits provided by ecosystem and hydrological services (Stanton et al., 2010). Landell-Mills and Porras (2002) also stress the potentially valuable role of PHES schemes in supporting coordination "by offering a transparent and efficient way of managing participants' interaction, a mechanism for sharing benefits to secure broad participation, and a mechanism for self-financing".

PHES schemes play an important role for the establishment of commonly agreed institutions (e.g. rules for land use) and also for the development of organizational structures. According to Landell-Mills and Porras (2002), PHES schemes have frequently spurred the creation of new organizations. Instead of posing threats to existing organizational arrangements, PHES schemes have complemented and strengthened existing hierarchical and cooperative structures (Landell-Mills and Porras, 2002).

In the context of their global review of PHES schemes, Lin and Nakamura (2012) stress the unique governance characteristics of PHES schemes in including intermediary organizations which "create a societal opportunity for enhancing ecosystem functions through mutual economic willingness-to-pay and willingness-to-accept payments between downstream and upstream stakeholders". Moreover, they highlight this *connecting* function of intermediaries as a critical starting point for integrated management. Actually, intermediaries often fulfill similar functions as RBCs but enjoy trust from all stakeholders since they act as brokers in their interest. This is a significant advantage of the PHES instrument compared to other instruments commonly applied as mandatory top-down approaches of political or membership linkages. The basic interest of an intermediary in a PHES scheme is to involve as many stakeholders as possible based on functional (bio-physical) linkages and engage them in the form of contractual agreements. In order to increase trust among stakeholders in organizational structures often provided by intermediaries, stakeholder representatives participate in the decision-making of these organizations. This active stakeholder involvement and participation is another advantage of local PHES schemes.

Implementation step	Efficiency consideration	IWRM context
Bio-physical cause-effect relationships between land use and service provision	Inclusion of potential provisioning areas limited to those areas with high service potential and low opportunity costs	Broad consideration of land uses having an impact on water resources
Socio-economic valuation of provision of and benefits from ecosystem services	Concentration on few service providers with largest service providing areas	Consideration of benefits from hydrological ecosystem services beyond those appreciated by potential buyers
Negotiation between stakeholders towards agreements on land use	Direct negotiations between buyer and sellers with individual agreements on actions and payments	Negotiating process as a broad forum to exchange different positions and participatory decision-making in a policy mix context
Continuous realization of agreed actions and payments	Conditionality on service provision (output-based)	Flexibility in additionality and conditionality according to commonly agreed goals

Table 5.13: Comparison of PHES implementation steps under efficiency optimizing consideration and in the context of IWRM (Author's work)

Besides directly addressing the problems of fit and interplay, PHES schemes have to be considered in the context of operational constraints described as information, capacity and funding gaps in Section 3.3. The bio-physical assessment and the economic valuation of service provision as preparatory elements for a PHES scheme can potentially contribute to closing information gaps at the level of IWRM operationalization. If conditionality and additionality are agreed principles, then strong incentives are provided to measure and monitor land use and service provision. Furthermore, the local assessment and negotiation of hydrological service provision based on defined land uses has certainly capacity building elements for local actors. Moreover, when PHES participants are involved in decision-making processes, important management skills are trained as well. Thus, the capacity gap is also addressed by PHES schemes in facilitating a form of co-management by PHES participants where local capacities and knowledge are included, e.g. on the opportunity costs of land use changes or their impacts on ecological functioning. Finally, PHES schemes, as originally intended, contribute to financing of water resources management by tapping local resources. Hence, PHES schemes are potentially able to co-finance IWRM operationalization through payments from beneficiaries of hydrological ecosystem services. This could reduce the funding gap of IWRM operationalization and incorporate resource costs into water prices (as proposed, for instance, by the EU-WFD).

Some aspects of PHES implementation may compromise PHES efficiency in straight economical terms of conservation but are beneficial in the context of IWRM or a broader development context. For instance, in order to increase the efficiency of PHES, only service providers with the lowest opportunity costs and the highest expected (relative) service provision should be considered (achieving environmental goals at the lowest cost). For the purpose of IWRM implementation, it is useful to involve all service providers, i.e. stakeholders, even if there are no beneficiaries readily identified. Thus, the IWRM objectives of involving all potential stakeholders with externalities may cause higher transaction costs (e.g. for additional capacity building and cooperation) which could be unfeasible under plain efficiency considerations. A stronger involvement of intermediaries also implies additional transaction costs, but a stronger involvement of local governments as intermediaries may be justified because of IWRM subsidiarity objectives and public good characteristics of services. Moreover, this provides opportunities for co-managing and co-financing. Table 5.13 summarizes the principal implementation steps of PHES schemes under primarily efficiency optimizing considerations compared to implementation in the context of IWRM.

Considering PHES implementation in an IWRM context seems reasonable since several operational tasks are equivalent. However, in an IWRM context PHES implementation would be required to be broader, thus, implying additional transaction costs. For instance, bio-physical assessments may not be limited to potential provisioning areas for actual hydrological services demanded but should include principal sustainable use considerations. This could lead to a more comprehensive land use planning including areas which are not of particular interest in a plain PHES context.

Considering PHES beyond individual market transaction and recognizing the role of local governments in caring for externalities that effect society as a whole (e.g. those beneficiaries of ecosystem services now and in the future which cannot be easily identified and engaged as user, i.e. payers) allows regarding them as instruments for co-management and co-financing of water resources management (cf., Gutman, 2003). Hence, local governments and other intermediaries may share the costs with ecosystem services buyers. Moreover, a redistribution of costs between up- and downstream parties may result in a more equal distribution according to perceived benefits (Reis et al., 2007).

Hydrological service	Examples for benefits	Benefit characteristics	Benefit type
Water flow regulation	Maintenance of dry season flows for water supply, flood control	Non-exclusive, non-rival	Public good
Erosion / sedimentation control	Reduction of landslides risk or reservoir sedimentation	Exclusive or non-exclusive, non-rival *Depends on characteristics of the resource system and beneficiaries*	Public or club good
Improved water quality	Nutrient and chemical load control, salinity control	Exclusive or non-exclusive, non-rival *Depends on characteristics of the resource system and beneficiaries*	Public or club good
Increased water supply	Groundwater recharge, increased available surface water	Non-exclusive, rival	Public good or common pool resource

Table 5.14: Benefit characteristics and type of hydrological ecosystem services; based on Reis et al. (2007)

Moreover, as documented by the common implementation practice of PHES schemes, ecosystem service beneficiaries and providers are involved in the policy process and can actively influence the outcomes. However, considering the multitude of ecosystem services in healthy ecosystems, it is impossible to engage all potential beneficiaries as ecosystem service buyers unless there is an obligatory fee that applies to all. While those who benefit most, e.g. major water users in a river basin, may possibly be engaged as service buyers there remains sufficient rationale for public authorities to engage as service buyers on behalf of the less easily identifiable beneficiaries of the present and the future. This explicitly includes society's benefits from non-use values. In this sense public authorities comply with their role as stewards for sustainable natural resource management. Moreover, public authorities as stewards or buyers of currently non-use values are in line with the idea of safe minimum standards and the precautionary principle as a central element of environmental policy. Public authorities may also assume responsibility for those ecological values which may not have been properly expressed, e.g. as stated preferences, in a valuation process.

Private beneficiaries and public authorities as beneficiaries on behalf of the society, acting as ecosystem service buyers reassigns traditional roles of natural resource management. In the traditional context of command and control regulation, public authorities design and set the formal rules that private actors have to comply with, thus, roles are assigned for the regulator and those being regulated, i.e. for those who actively define the policies for what is right and wrong and those that have to respond passively. In contrast, in a situation where public authorities and private actors are taking similar roles as ecosystem service buyers there is much more room for mutual policy design, since traditional hierarchies are offset.

Besides being a buyer of ecosystem services, public authorities are often important facilitators of knowledge on laws or improved practices, acting as coordinators between different buyers and network agents in connecting different public and private parties, e.g. involving other governmental organizations from different policy sectors (Tognetti et al., 2006). This way they have potential to improve stakeholder involvement and public participation in the policy design and implementation process. Public authorities (i.e. local governments), thus, can promote co-management and co-financing for IWRM. In this context, Reis et al. (2007) propose to determine the degree of public authority involvement in policy mixes containing PHES schemes based on the public good characteristics of hydrological ecosystem services. According to the characteristics of excludability and rivalry of benefits from hydrological ecosystem services illustrated in Table 5.14, three general categories can be identified (cf. Reis et al., 2007).

One category encompasses hydrological with public good character, for instance flood protection. Public good services are non-rival and non-exclusive as it is practically impossible to exclude anyone from the benefits. For payment schemes addressing this kind of services it seems reasonable if a public authority provides incentives for upstream communities to supply the service on behalf of a larger benefiting society (cf. Reis et al., 2007). However, by making public goods exclusive they may be transformed to toll or club goods. This second category of services incorporates hydrological services related to improved water quality, for instance reduced sedimentation or reduced flow of nutrients, chemicals or salinity. These services can be made exclusive through regulating the access to the water body, while they will always remain non-rival. Here again, a payment scheme seems suitable where a government agency pays upstream providers and regulates downstream access to the water body by setting prices. The service automatically becomes exclusive in cases where only one private beneficiary, e.g. a hydro power company, is present. In this case direct payments often work well without the need for downstream institution building. Reis et al. (2007) argue that in the case of multiple beneficiaries, e.g. irrigation systems, user payments could also work if the community is able to exclude community members that do not pay even if the use is non-rival. Finally, Reis et al. (2007) introduce hydrological services with common-pool resource characteristics of rivalry and non-excludability. This category includes all services related to increased quantitative supply. Contrary to public good services, if access to the service is made exclusive on the buyer's side, these services can be traded. According to Ostrom (1990), three prerequisites have to be met in order to exclude service beneficiaries who do not contribute to its supply:

- Clearly defined boundaries: Clear definition of who has the right to utilize the service, and clear definition of the boundaries of the common pool resource itself
- Monitoring
- Graduated sanctions

According to Reis et al. (2007), individual service users "will only be able to reach excludability if they establish an institution that secures clearly defined boundaries and adequate tools for monitoring and sanctioning, i.e. if they cooperate". However, irrespective of the kind of hydrological service category cooperation among providers is often necessary due to the fact that the service usually cannot be provided by a few single landholders (Adhikari and Boag, 2013). This is especially true for developing countries where providers will most likely be smallholders and a measurable environmental benefit for downstream communities will only be ensured if they collaborate.

The categorization of hydrological services with regard to excludability and rivalry characteristics by Reis et al. (2007) provides some useful reasoning whether a PHES scheme may be based on a direct payment mechanism between upstream and downstream individuals / communities / corporations, and to which extent the action of government agencies is advisable. Since several hydrological services can feature public good characteristics, public authorities acting as service buyers or at least contributing to payments seems reasonable. Some non-rival services can be made exclusive or are practically exclusive, thus, governmental involvement as a buyer may not be necessary. However, PHES schemes for rival and non-exclusive hydrological service, according to Reis et al. (2007), rely heavily on the level of cooperation on the beneficiaries' side. While in the context of payments for a private good competition is the main driver, here cooperation is needed to establish a payment scheme. Thus, as PHES schemes are based on cooperation and not on competition they may not be considered as actually market-based.

Consequently, Reis et al. (2007) point out that "the key question is therefore not, whether markets should be introduced instead of cooperative institutions or government regulations, as they do not represent an alternative for the management of water resources. The question is rather, whether market-like mechanisms are an appropriate tool for resolving the problem of unequal cost and benefit sharing in watershed management within a certain institutional framework based on cooperation or public regulation". According to the authors, PHES schemes may also be considered as a means to promote cooperation among water users where there are no institutions for common-pool resources management or where such institutions do not work properly.

Considering the above discussion of property rights and public good characteristics it becomes clear that there is often also a public or common good concern which justifies the involvement of governmental actors in PHES schemes. Participation of public authorities in PHES schemes means involvement of those establishing (formal) rules and those being affected by the rules. Here an important characteristic of PHES schemes is that participants can influence the rule making process, while existing formal but also informal rules may be considered. This allows for the flexibility required to meet specific local contexts.

Although considered merely an economic instrument, PHES are characterized by communicative and informative instrumental aspects as well as a strong interaction with command and control mechanisms. In fact, PHES can induce a discussion on how ecosystem services should be provided, e.g. through compensation or legally enforced protection. As a result of the PHES establishment different service providing areas and suitable uses are identified. This information represents, on the one hand, a baseline scenario for future ecosystem service improvements and also the basis for land use planning based on river basins, which is a major element of river basin management plans usually used to implement IWRM. This requirement of the PHES implementation process, thus, can easily serve the purpose of IWRM operationalization. Furthermore, areas which are most vulnerable for hydrological ecosystem service provision (e.g. spring areas, riparian areas and steep slopes) are in most cases protected by law but lack of enforcement has still lead to degradation. Here, local PHES have the potential to contribute to an improved enforcement of the existing rules based on the communication aspects of PHES and the provision of material, e.g. to build fences around these areas. The benefits from IWRM implementation are often not tangible right away, while PHES can demonstrate benefits and trade-offs in a demonstrative manner. Another aspect is the funding needed for IWRM implementation which in most cased is provided by federal subsidies. This creates dependencies at lower governmental levels and funding is often insufficient or lacking. PHES can contribute partly to IWRM funding through tapping of additional financing sources if elements of PHES implementation are also used for IWRM purposes, e.g. land use zoning or baseline assessment and monitoring. Thus, PHES offer the possibility to facilitate context-specific integration based on solving the problems of fit and interplay in an interdependent manner. The relationship of ecosystem service providers and users can possibly define the degree of integration (spatial and sectoral) based on related transaction costs.

5.4 Summary

PHES originally evolved as market-based instruments in order to improve efficiency in environmental conservation. However, as different economic conceptualizations have shown, the instrument provides advantages over traditional regulation instruments. Practical application of PHES schemes reveals that stakeholder interaction and finding mutual agreements seem to receive more attention than strictly market-based efficiency consideration. This is especially true for local schemes where stakeholders, i.e. most importantly service providers and buyers, come to agreements on land uses and context-specific definition of conditionaliy and additionality. Hence, local PHES have the potential to improve stakeholder involvement and public participation in policy design and implementation in water resources management. Furthermore, incentives provide the motivation for stakeholder actions and introduce behavioral change by involving those that actually have a stake in the local IWRM process in order to identify the right scale for management based on fit and interplay.

There are several steps inherent to the PHES implementation process that are equally important for IWRM operationalization: communication of the river basin as the management unit, involvement of up- and downstream stakeholders, data acquisition of water and land uses, identification of hydrological cause-effect relationships, exchange of value perceptions and preferences. These implementation steps are mutually required and facilitate cooperation. The majority of steps to implement IWRM on the local level are often not taken yet (Akhmouch, 2012), but as evidence from case studies suggests, they are taken within PES projects when hydrological ecosystem services are considered. Moreover, these PHES schemes often provide the financial resources and a conceptually inherent necessity to carry out these steps which otherwise would possibly not be done. Thus, it can be argued that where PHES schemes are carried out, critical elements of the IWRM process used to be absent or at least lacking implementations. The documentation of many PHES case studies suggests that their implementation includes several tasks that are also part of the operationalization of IWRM, more specifically the elaboration of river basin management plans, for instance, land use planning on the basis of river basins. Furthermore, within PHES schemes organizational structures are established that fulfill similar tasks to those of river basin committees. However, compared to committees that are installed in a top-down manner the PHES organizations are more context-specific and based on particular functional linkages rather than political or membership linkages. It can be concluded that PHES schemes provide critical incentives to engage different stakeholder across sectors and administrative boundaries and to build up knowledge on SES needed for practical management. Thus, PHES may possibly be more than just an end of establishing economic incentives for conservation, but also a means for the operationalization of IWRM. This additional objective (i.e. side effect) of PHES may justify higher transaction costs and efficiency trade-offs.

The PHES implementation process offers several opportunities to overcome some principal implementation gaps of IWRM operationalization (see Section 3.3) like the information, capacity, and funding gap. For other implementation gaps, e.g. administrative gaps, it can be a suitable stepping stone. PHES schemes may also benefit from stronger involvement of public authorities and other networking intermediaries because they can reduce their share on transaction costs and provide additional motivations to engage service providers (e.g. those with more or less high opportunity costs but also high potential for service improvement). Public authorities can provide resources for capacity building and knowledge generation.

This chapter considered general characteristics of PHES and the specific implementation steps of local voluntary contractual agreements as a potentially suitable instrument to foster IWRM operationalization. In principle, these PHES schemes have the potential to address the problems of fit and interplay in order to find the most appropriate scale for management based on context-specific interactions of natural and human system's characteristics. Thus, spatial and temporal phenomena are being considered not only in the context of ecology but also in the context of social interactions. Moreover, both, PHES schemes and the IWRM operationalization process, may provide mutual benefits. However, documented difficulties and limitations of current PHES practices have to be addressed. This requires basing PHES schemes on clear and consensual scientific evidence, linking land uses to the provision of services, flexible, ongoing and open-ended contracts and payments, multiple sources of revenues which are sufficient and sustainable over time, monitoring of compliance, land use changes, and the provision of services; and the flexibility to allow adjustments to improve their effectiveness and efficiency in order to adapt to changing conditions (cf., Mayrand and Paquin, 2004). This way better evidence can be gathered from the beneficial impacts of sustainable land-management practices on water flow and quality, and on the ability of payments to influence the behavior of landholders. In achieving this, PHES may become an integral part of water resource management and allocation policy (Porras et al., 2008).

As the global review of PHES case studies has demonstrated, most experience and documentation on PHES schemes exists in Latin America. With respect to the land area and population covered there is a somewhat higher amount of cases in Central America than in South America. The largest number of local PHES schemes is located in Central America. The following chapter analyzes the process of IWRM implementation in Nicaragua and draws on a Nicaraguan case study evaluation of a Payments for Hydrological Ecosystem Services scheme for possible conclusions on the extent to which this and other schemes in Nicaragua may facilitate further IWRM operationalization on the river basin scale. Nicaragua is chosen, because although there is no national PES scheme several local PHES schemes have already been implemented. Moreover, the implementation of PHES has been accompanied by significant national efforts to implement IWRM. These are the reasons for choosing Nicaragua as an empirical case study example in order to assess the potential role of local PHES in the context of IWRM operationalization.

6 Contributions of Payments for Hydrological Ecosystem Services to IWRM in Nicaragua

The role of PHES in the context of the IWRM implementation process refers to user-financed publically and/or privately organized PHES. Based on the findings of the previous chapter, it is assumed that these user-financed PHES could play an additional role as part of a bottom-up strategy for IWRM implementation due to their functional role and realization procedure. In order to analyze the role of PHES schemes in an empirical context, Nicaragua is chosen as a reference country. In Nicaragua, the IWRM process is initiated at the national level but still lagging behind in operationalization. However, predominantly local user-financed PHES are being established at the same time.

This chapter describes the IWRM implementation process in Nicaragua until today. It briefly analyzes the principal implementation gaps and operationalization constraints against IWRM implementation. The implementation process of the PHES scheme in the Gíl González river basin is documented and used as a case study to demonstrate the general findings of the potential of PHES to contribute to the operationalization of IWRM as discussed in the previous chapters. While the Gíl Gonzáles case is used to provide details of the implementation process other experiences of local PHES schemes in Nicaragua are also frequently referred to. Together the schemes form the basis for an analysis of the contribution local PHES schemes make to the operationalization process of IWRM in the context of a broader policy mix.

6.1 The process of Integrated Water Resources Management in Nicaragua

The legislative regulation of the use of water resources in Nicaragua began in 1904 with the establishment of the Civil Code. During this time the government's priority was to regulate the access to water resources in order to promote agricultural activities. Therefore, the Civil Code promoted the private management of water and other natural resources through specific laws based on the ownership of land. In 1919 and 1923 the use of superficial water resources for the generation of energy was regulated by governmental decree. Through this decree the central government formulated the requirement for concessions for the use of public waters by the executive power. The General Law on Exploitation of Natural Resources was approved in 1958 which included water as a natural resource of state property whose use shall be regulated by a specific law. In 1969 a governmental decree established procedures for the control of agricultural water use through the National Index of Wells. However, the use of this index was interrupted during the political revolution in the 1980s which has led to the use of water basically free of charge in small- and large-scale irrigation (cf., Gómez et al., 2012). The index of drilled wells has recently been reactivated by the Ministry of Industry and Commerce. However, during the 1980s the promulgation of the National Constitution marked an important step for the country's environmental regulation. The constitution establishes the authority of the state as the custodian and administrator of all national natural resources. Moreover, the constitution creates two institutions, which even today have important functions for water management: the Nicaraguan Institute of Natural Resources, now called Ministry of the Environment and Natural Resources (Ministerio del Ambiente y los Recursos Naturales; MARENA), and the Nicaraguan Institute for Territorial Studies (Instituto Nicaragüense de Estudios Territoriales; INETER). The evolution of IWRM in Nicaragua has its origins partly in the 1980s when the Municipality Law (Law No. 40: Ley de Municipios, 1988) was issued, which initiated the decentralization of natural resources management by passing over responsibility and household budget from the central government to the local municipalities.

The Municipality Law was reformed in 1997. With this reform the municipal activity in the areas of policies, administration and finances was newly regulated. The reformed law transfers the competence for all areas related to the municipality's socio-economic development as well as the conservation of the environment and natural resources within their territory. The financial resources to execute these competences, according to the law, originate form own revenues as well as fiscal transfers from the central government. Other competences established in the Municipality Law refer to the planning, standardization and control of the use of the soil as well as urban, peri-urban and rural development (Art. 7). These competences empower municipalities to make land use planning decisions within their territory. Thus, local governments are able to define specific areas, e.g. for urban settlements, agricultural use and nature conservation. Additionally, municipalities are empowered by the law to develop, conserve and control the rational use of the environment and natural resources as the basis for the sustainable development of the municipality by forming initiatives in these areas and contributing to their monitoring, supervision and control in coordination with corresponding national entities (Art. 7). This allows municipalities, through their councils, to legislate the protection and conservation of the environment and natural resources located in their territory and to coordinate activities with corresponding governmental agencies at higher political levels. These responsibilities turn municipalities into important actors for IWRM operationalization at the management level.

Finally, the law states that the provision of basic water, sanitation, and electricity services to the population shall be a municipal responsibility. Besides the assignment of this responsibility, the law remains unclear regarding the transfer of assets from the central government. However, according to Walker and Velasquez (1999) direct service provision by the municipalities is not a general practice in Nicaragua. The municipalities of Matagalpa and Jinotega, for instance, where the Nicaraguan Institute of Aqueducts and Sewers (INAA; national regulation agency for water supply and sewer systems) delegated administration to the local authorities in 1992, following local government pressure, are some of the few successful exceptions. In most other cases, municipal water supply is provided by the state owned company ENACAL being the principal national operator of water supply and sewer systems.

The politics of the 1990s were strongly influence by external factors and international developments in the environmental sector. Policies with tendencies to reduce the role of the central state and to promote more participation of the private sector and the civil society as well as a general process of decentralization responded to the establishment of the Dublin Principles and the international agreements of the World Summit of Rio de Janeiro both in 1992 (see Section 2.1). Consequently, in 1994, the Government of Nicaragua introduced sustainability criteria for the use and management of natural resources, the Environmental Action Plan. A core theme of the plan was the formulation of a national policy and associated strategies for the rational development and management of water resources. Following the Environmental Action Plan, the Nicaraguan and Danish Governments agreed on the implementation of a Nicaraguan Water Action Plan. A Danish Consortium of consultants was contracted to carry out the Nicaraguan Water Action Plan in cooperation with local experts. The Action Plan is considered the first step towards integrated water management through the development of policies, legislation, institutional adjustments and technical solutions to identified problems (Milton, 1998). Among the results of the Water Action Plan presented in 1998 were proposals for a National Water Resources Policy and a draft for a National Water Law (Ley General de Aguas Nacionales). This represented an important input to the promulgation of the first National Policy on Water Resources (Política Nacional de los Recursos Hídricos, Decree 107-2001) in 2001. This policy considers the integrated management of water resources as a principal management paradigm. Several additional laws followed addressing the management of forest resources and the violation of environmental protection among others. According to Milton (1998), a key recommendation of the action plan was "the establishment of an independent and neutral water authority, recognizing the continuing contribution of the National Water Resources Commission, and the need to harmonize and put into operation its functions". Moreover the Action Plan identified the following major obstacles to the implementation of IWRM in Nicaragua at this time:

- Indecision and lack of political will to press on with institutional changes
- Budget and financial restrictions on the part of the institutions involved in water management processes
- A weak institutional structure
- Lack of definition of functions and responsibilities of organizations involved
- Institutional coordination almost non-existent
- Scarcity of technological resources
- A notable lack of human resources training in the sector
- Insufficient monitoring and control of water quality
- A notable lack of basic knowledge on the availability and condition of the resource

However, major recommendations of the Water Action Plan on actions toward IWRM implementation have not been implemented by the Nicaraguan parties. According to Milton (1998), this "indicates the lack of ownership and driving force of the project at the national level". Nevertheless, the Water Action Plan was prepared in an attempt to translate the guiding principles of the Dublin-Rio process into a national water resources strategy. The National Commission of Water Resources (Comisión Nacional de Recursos Hídricos, CNRH), an inter-institutional body created in 1983 composed of the heads of six state agencies concerned with water management and members of the National Assembly, was reactivated and reorganized during the implementation of the action plan. According to Marisa (2000), the principal outcomes were "the recognition of the need for a unique water authority to regulate and administer water resources in quantity and quality, a coherent water policy, a proposal for a comprehensive water law, and an action program to implement the required changes". Hence, the action plan contributed to the definition of requirements for integrated water resources use and management.

However, the action plan already recognized that creating a unique authority to regulate water resources in quality and quantity could result in conflicts with the existing Ministry of the Environment and Natural Resources. Furthermore, Marisa (2000) argues that the action plan still follows a top-down approach where users are perceived as passive rather than active actors. Although the participation of non-governmental stakeholders in the execution and management phases is recognized as important, emphasis is put on governmental actors.

In 1996 the General Law on the Environment and Natural Resources (Law No. 217: Ley General del Medio Ambiente y los Recursos Naturales), for the first time in a legal context, referred to river basins as the appropriate spatial unit for management and use required to guarantee sustainability of water resources. Moreover, the law declared water resources as a public domain and reserves the property of coasts, river channels and lakes as well as the riparian area within 30 meters from the maximum river, lake and ground water levels for the state. The General Law on the Environment and Natural Resources is the main regulatory instrument for water source protection and pollution control. The law allows creating of a National Environmental Commission and a Special Procurator for the Environment and Natural Resources. It gives the Nicaraguan Institute for Territorial Studies (INETER) and the Ministry of the Environment and Natural Resources (MARENA) powers to organize the environmental and territorial structure of the country. Furthermore, the law creates the National Environment Fund and establishes the National Environmental Information System (Walker and Velasquez, 1999). The General Law on the Environment and Natural Resources recognizes the municipalities as the main actors in environmental regulation within their territory and established, in a broader sense than the Municipality Law alone, the fields of action of municipal authorities with respect to environmental regulation.

The Nicaraguan environmental policy, initiated with the General Law on the Environment and Natural Resources, basically relies on command and control instruments which consist of quality standards for effluents, environmental permits for new investment projects, and the regulation of the use of pesticides and other toxic substances (Nuñez Ferrara, 1998). Due to implementation difficulties, these instruments have neither been very effective in reducing pollution nor in controlling the use of natural resources, including water (cf., Lutz et al., 1994; Chacón and Pratt, 1996; Pagiola et al., 2002). Implementation difficulties stem from the lack of appropriate financial resources assigned to the Ministry of the Environment and Natural Resources for enough trained staff and the financial resources to monitor and enforce regulations (Nuñez Ferrara, 1998; BCN, 2000). Furthermore, the ministry does not have the capacity to penalize those who do not comply with the norms (Marisa, 2000). According to Nuñez Ferrara (1998),in 1999 the personnel allocated to environmental protection by MARENA counted only 22 persons at the central national level and 31 in the rest of the country. Moreover, the environmental command and control instruments implemented at the end of the 1990s were not accompanied by economic incentives, which has further restrained compliance (cf., Marisa, 2000). Although the need for economic incentives to promote the reduction of waste and waste water at the source was discussed during the drafting of the General Law on the Environment and National Resources, this was not followed by specific instrument development.

Finally, in 2007 almost a decade after the first draft in 1998, the first National Water Law (Law 620: Ley General de Aguas Nacionales) was passed. The National Water Law establishes the general framework for IWRM policies in Nicaragua including all international principles. Moreover, with the new water law, the three key IWRM change areas of the Global Water Partnership (GWP) for the adaptation of IWRM through (1) an enabling environment, (2) assignment of institutional roles, and (3) appropriate management instruments were addressed and recognized (see Section 2.1). The law pursues the objective to establish the institutional and juridical framework for the management, conservation, development, use of all water resources of the country (superficial, subterranean, residual or of any other nature) in a sustainable and equitable manner. At the same time it guarantees the protection of other natural resources, ecosystems and the environment in general. The law puts emphasis on the integrated management of the resource based on river basins, sub-basins and micro-basins. The use of the resource for human consumption is granted as a priority before agricultural, ecological, industrial and other uses. The guiding principles of the National Water Law are:

- Water is a strategic resource of national priority
- Knowledge on hydrological processes is a prerequisite for the effective management of water resources. The state provides all necessary resources to gather sufficient information
- The preservation and protection of water resources is a fundamental task of the state
- Responsible management and the provision of water resources for human consumption is of highest national priority
- The management of water resources is based on the integrated management of surface and sub-surface water resources, their multiple uses and the interrelationships of different ecological media (water, air, soil, flora, fauna and biodiversity)
- Public participation
- Polluter Pays Principle for those who pollute and compensations for those who use efficiently and cleanly
- Precautionary principle
- Subsidiarity principle (ANA, 2010)

According to the National Water Law, water resources planning implies the elaboration of a National Water Resources Plan through the National Water Authority (Autoridad Nacional de Agua, ANA) as the basis for River Basin Management Plans and Programs executed by RBOs. River Basin Management Plans have to be approved by the National Water Resources Council (Consejo Nacional de Recursos Hídircos, CNRH). These plans form an integral part of the planning process of water resources. The CNRH periodically evaluates the implementation advancements of the National Water Resources Plan and the River Basin Management Plans. The National Water Resources Plan and the River Basin Management Plans are published in the governmental communication tool La Gaceta.

The National Council for Water Resources is (re)created (based on the former National Commission of Water Resources) by the National Water Law as the ultimate instance of the IWRM framework representing the principal normative and highest strategic policy level for water resources management. The National Council for Water Resources is composed of representatives of the Ministry of Environment and Natural resources (MARENA), the Ministry for Agriculture and

Forestry (MAGFOR), the Ministry of Health (MINSA), the Ministry of Industry and Commerce (MIFIC), the National Institute for Territorial Studies (INETER), the administration of water supply and sewage, the administration of energy, a representative of the Ministry of Energy and Mining, a representative of the National Commission for Water Supply and Sewage (Comisión Nacional de Agua Potable y Alcantarillado Sanitario, CONAPAS), one representative of each of the Regional Councils Autonomous Regions of the Atlantic Coast, four representatives of the productive sectors, and four representatives of user organizations. Through this multi-sectoral and multi-stakeholder composition the council is designed to be a forum for consultation and participation at the highest political level. Moreover, the council represents a consulting and cooperation facilitating organ with the purpose to approve the general politics in the context of water resources and to supervise the tasks executed by the ANA. Thus, according to the National Water Law, the specific functions of the CNRH are:

- Elaborating and updating the National Water Resources Policy
- Approving the National Plan for Water Resources as well as the plans and programs for river basins
- Providing consultancy and inter-sectoral coordination for the integrated management and planning of water resources
- Resolving committed issues concerning water management as well as revenues and income of the National Water Authority (ANA)
- Approving the establishment of RBOs and River Basin Committees (RBC)
- Approving concessions (after consultation of all sectors and actors involved) for strategic water uses or those which involve more than one sector, river basin or requiring the construction of large size hydraulic works
- Approving its internal regulation

Hence, the principal responsibilities of the CNRH relate to strategic tasks concerning approval and supervision of national water politics.

The CNRH is an attempt to coordinate the different water-related tasks of existing governmental institutes and ministries at the national level in order to concentrate the authority to regulate water in both quantity and quality. The principal water-related governmental institutes and ministries and their respective tasks which are supposed to be coordinated by the CNRH are:

- The Ministry of the Environment and Natural Resources (MARENA) in charge of preparing protection plans for natural resources and control of water quality.
- The Ministry of agriculture and forestry (MAG-FOR) responsible for the development of agriculture activities, including industries, forestry, irrigation, and control of agrochemicals as well as the establishment of protection plans for soil and water resources.
- The Ministry of Health (MINSA) controlling aspects of water supply and sanitation, water quality, and agrochemicals.
- The National Institute for Territorial Studies (INETER) in charge of the physical land use planning, inventories and characterization of water resources.
- The Ministry of Energy and Mining (MEM) regulating the use of water for hydro power and geothermal activities.

This illustrates that several of these organizations have overlapping mandates regarding the protection of water resources quality and quantity. Their sectoral vision in planning and administration, seldom considering the effects on other users and on the environment, has hindered cross-sectoral cooperation and resulted in ineffective management of water resources (cf., Marisa, 2000). Consequently, several of the tasks of the water-related institutes and ministries have been reassigned to the National Water Authority through the National Water Law in order to improve coherence in water policy. However, the CNRH remains the highest instance for water resources management and approves the principal activities of the ANA.

In order to realize an Integrated River Basin Management (IRBM), the National Water Authority has been empowered by the National Water Law to establish River Basin Management Plans and to submit them for approval to the CNRH. Further tasks of the ANA are to supervise the water use in river basins and to grant authorization and concessions for water use of any kind (Baca et al., 2012). According to the National Water Law, ANA carries out the following technical-normative functions:

- Formulating and elaborating the National Plan of Water Resources
- Coordinating the elaboration of Water Resources Plans for river basins and supervising compliance
- Elaborating hydrological balances for river basins in coordination with competent authorities
- Proposing regulations for river basin management, including aquifers
- Realizing the characterization of water bodies for their potential uses
- Proposal of declarations of zones of protection, nature reserves and bans for hunting to competent authorities
- Granting, modifying, extending, suspending or terminating the concession of titles and licenses for water use as well as permits for the dumping of sewage into water bodies of the public domain
- Regularly conducting studies and analysis on economic and financial valuation according to supply source, location and type of use, that support the criteria for the collection of water rates and charges, including payment for hydrological ecosystem services

- Proposing classification declarations of high-risk areas for flooding

Thus, ANA regulates, administers and supervises the management of water resources at the national level in all aspects. Besides technical-normative functions ANA must also comply with the following technical-operational functions:

- Managing and supervising in an integral manner and on the basis of river basins, all national water resources and preserving and controlling their quantity and quality. Elaborating the management plans of different aquatic ecosystems together with MARENA and municipal councils concerned
- Managing and supervising public hydraulic works of the state
- Establishing, organizing and managing the Public Registry of Water Use Rights
- Organizing and coordinating the National Water Resources Information System in order to determine the availability of water in quantitative and qualitative terms as well as to identify its use and users
- Constructing public hydraulic works or contracting third parties for construction
- Conflict resolution of water uses
- Formulating and applying programs to implement volumetric measurements of all water uses
- Defining requirements and alignments to establish of districts and units of irrigation and drainage
- Executing additional and transient technical-operational functions of River Basin Organizations as defined by the National Water Law
- Acting as a review body for decisions made by River Basin Organizations

The National Water Law intends to establish River Basin Organizations as dependent derivatives of the ANA at the level of the 21 principal national river basins. These RBOs function as governmental bodies with technical, operational, administrative and legal functions, coordinated and harmonized with the ANA in order to manage, control and supervise the use of water resources at their corresponding river basins. If there is no RBO, the ANA will immediately execute all technical and operational functions on a temporary basis until a RBO has been assigned. The establishment of RBOs represents the principal mechanism of decentralization provided by the National Water Law.

The RBOs are to be composed of a directive council, a director as well as administrative and technical entities. The council incorporates a delegate of the ANA chairing the council, all mayors of all municipalities forming part of the river basin as well as a delegate of INETER, MAGFOR, MINSA and MARENA. The council may invite additional parties, e.g. representatives of water user associations or from other governmental authorities, if this is considered convenient. The directors of the RBOs are assigned by the National Council for Water Resources and proposed by the ANA.

River Basin (Sub-Basin or Micro-Basin) Committees (RBC) are created by the National Water Law to promote the participation of the civil society in the IWRM process. The establishment of River Basin, Sub-Basin and Mirco-Basin Committees shall be promoted by the respective RBO on the basis of the spatial dimensions, the quantity of water resources and the diversity of water uses within the river basin. The ANA establishes mechanisms for the RBOs that promote the formation of River Basin Committees and their approval through the CNRH. These committees are composed of representatives of all water users of the different water uses within the river basin, representatives of the directive council of the corresponding RBO, representatives of the regional autonomous council (in case of the autonomous regions of the Caribbean Coast of the country) and representatives of accredited NGOs. RBCs shall consist of groups with an equal number of representatives of water users, the civil society and governmental actors.

The committees represent fora for consultation, coordination and agreements between the RBOs, governmental entities, municipalities, NGOs and the water users of the respective river basin. The committees participate in the formulation of plans and programs which are elaborated by the RBOs or the ANA and are supposed to safeguard better administration of water, the development of hydraulic infrastructure and their services, the management of financing mechanisms which allow supporting activities above the conservation and protection of water resources. Plans and programs of lower order committees have to respect the frameworks provided by higher order committees, RBO or the ANA (ANA, 2010). Thus, RBCs provide the basic means of public participation in the IWRM process. They represent the only organizational elements where (local) non-governmental actors, e.g. water users, may be included.

Figure 6.1 illustrates the organizational framework for IWRM in Nicaragua as established by the National Water Law. The arrow on the left indicates the decision-making power of the different organizations and their respective level of influence in policy making. Although organizational structures are provided for the normative, strategic as well as for the operational level and mechanisms for decentralization and public participation are included, the decision-making power and influence on policy making is concentrated at the national level.

The IWRM process in Nicaragua is now at a typical stage of implementation (see Section 2.1.2). Several important steps for the establishment of an enabling environment and the assignment of institutional (i.e. organizational) roles, for instance through the National Water Law and the creation of the CNRH and the ANA, have taken place at the national level and subsequent steps of decentralization and the application of management instruments at the operational level are designated by law and regulations. However, in the context of the Nicaraguan IWRM process the emphasis is placed on applying the IWRM concept at the constitutional and associative central governmental levels rather than the operational

Figure 6.1: Illustration of the organizational framework for IWRM in Nicaragua (Author's work)

level and this is what García (2008) has identified as "major hurdles" of IWRM implementation in Latin America. As a result, García (2008) argues that "the achievement of institutional reforms proved to be extremely difficult in practice, especially those related to the water allocation among competing uses function". This is true for Nicaragua. Hence, the Nicaraguan IWRM process encounters the typical problems of IWRM implementation and operationalization just as many other countries do (see Section 2.3). The problems of institutional fit and interplay are addressed in a more ore less "standard" fashion without specific mechanisms to recognize site-specific problem contexts. Moreover, Nicaragua struggles with the principal operational constraints as discussed in Section 3.3. These implementation gaps of IWRM in Nicaragua are discussed in detail in the following section in order to illustrate the specific problems involved in the operationalization of IWRM.

6.1.1 Implementation gaps

The implementation process of IWRM in Nicaragua as described in the previous section dates back to the 1990s when the topic was strongly promoted by the international (donor) community. Since then, it has been a long way towards the first steps of realization. The National Water Law, a major milestone for the IWRM process, required almost a decade to be passed. Moreover, it took another 3 years to establish the organizational framework, most importantly the National Water Authority (ANA), and to appoint a director. And this is only the beginning, other important organizations such as the 21 RBOs for the mayor river basins of the country and possible river basin committees at the sub- and mirco-basin level are still not in place which also applies to several regulations of instruments announced by the National Water Law such as the Public Registry of Water Use Rights. Thus, the initial IWRM process in Nicaragua can be characterized as lengthy at least.

At the moment the ANA is elaborating a new National Policy on Water Resources (PNRH) because several weaknesses of the national water policy from 2001 impede the proper functioning of the ANA (Morales, 2012). Since the National Policy on Water Resources represents the *master instrument* of the national IWRM process, it is supposed to provide strategic orientation to all other IWRM related instruments. Without a new National Policy on Water Resources none of the other steps, e.g. the creation of RBOs and River Basin Committees, will advance.

Although the National Water Law has some delegating elements and presents itself as a decentralized organ, in reality it concentrates the IWRM process at the national strategic policy level in pursuing a top-down approach towards IWRM implementation. The decision-making process of all organizations introduced is determined from top-down and principally based on central governmental power (as illustrated in Figure 6.1). While municipalities are at least included in RBOs, representatives of the civil society are not. The planning and management of water resources is supposed to be realized through the ANA at the national level by means of the National Policy on Water Resources. The elaboration of River Basin Management Plans is delegated by the ANA to the RBOs. However, the national plan precedes the river basin plans reducing the decision-making power at lower levels (Novo and Garrido, 2010). Moreover, the regulatory decree of the National Water Law states that RBOs will be installed if financial and human resources of the ANA permit this. The financial resources for the establishment of RBOs and for the elaboration of the National Policy on Water Resources and River Basin Management Plans is supposed to be provided by the National Water Fund (Fondo Nacional del Agua). Furthermore, besides donations from development cooperation agencies, the fund will be alimented through revenues from water fees. For the introduction of water fees an additional law has to be passed. However, both the fund and the law for water fees have still to be created. Additionally, water fees water users and uses have to be identified first before water fees can be applied. This is meant to be achieved through the National Registry of Water Use Rights which also still needs to be established. As a result the limited financing of the ANA impedes the establishment of RBOs, leaving all responsibilities to the ANA. Hence, a central agency remains in charge of water resources planning for different river basins without defined proceedings for stakeholder involvement and public participation. Furthermore, the ANA proposes all regulations on how RBOs should manage water resources. These regulations determine how RBO carry out their functions, what organizational structures look like, how to finance them and all other questions related to the proper functioning of RBOs. Additionally, any ANA proposal must be approved by the CNRH.

Currently, the ANA is assuming all strategic and operational decision-making power without being able to delegate these functions, at least partly, to lower levels such as RBOs. As a result, the ANA has to set priorities in its tasks based on strategic considerations. For instance, at the moment, the ANA is focusing on the realization of technical studies (including delimitation of the principal river basins of the country) capacity building and investments in equipment to strengthen its institutional capacities for IWRM development. A further focus is on establishing the laws and regulations, e.g. the National Registry of Water Use Rights and the National Water Fund, which are still lacking fulfill the functions assigned to the ANA by the National Water Law. However, operational priorities of specific management context cannot be attended to because of the prevailing resource and capacity constraints.

However, apart from the struggles of assigning institutional roles and establishing further rules and laws or the proper functioning of the new water law, the IWRM process in Nicaragua addresses the principal implementation gaps of institutional fit and interplay. The mismatch between hydrological and administrative boundaries (the administrative gap conceptualized as problem of fit, see Section 3.1) has already been recognized as a problem by Nicaraguan politics in the 1990s and solving it through the establishment of river basins as management units was stated in the General Law on the Environment and Natural Resources in 1998. However, until the National Water Law had been passed this was merely a good intention. The new water law places strong emphasis on an integrated management of water resources according to the boundaries of river basins and calls for the creation of RBOs at the level of the country's major first order river basins as the organizational framework. However, the principal focus for solving the problem of fit is placed on the first order river basins, while the level of sub-basins and micro-basins is also taken into account with the respective establishment of sub-basin and micro-basin committees. However, the lower levels have much less weight in decision-making since their roles are limited to consultative ones without decision-making power. Moreover, it remains unclear how and by whom these committees will be established. Since all organizational structures are to be proposed and approved by upper level bodies, there is a strong top-down influence on IWRM implementation at present. The strongly pronounced hierarchical structures of strategic plans as the basis for river basin, sub-basin and micro-basin management plans illustrate this top-down approach. Thus, there is not yet an instrument in place that takes the specific natural context and related properties into account. On the contrary, the fundamental basis for all other management plans is the National Policy of Water Resources which still has not been passed. This is again exemplary for the general lack of institutions, organizations and specific laws that are needed for the proper functioning and operationalization of IWRM. Moreover, as long as RBOs and RBCs are not established, the ANA has to solve the problem of fit based on a national perspective and it is probably impossible to consider specific natural and social local contexts.

With regard to addressing problems of institutional interplay, the National Water Law actually includes important improvements. An attempt is made to bundle the formerly dispersed roles and sectoral functions of different ministries within the CNRH, in form of a coordinating organization. Thus, the new law has simplified and clarified the distribution of competencies and roles among different governmental entities at the national level (cf., Novo and Garrido, 2010). Improvement of horizontal interplay is achieved through the establishment of the CNRH, composed of formerly dispersed and sector-oriented ministries, water sector organizations as well as mayors, representatives of water users and the industry. Thus, horizontal interplay is provided at a normative and strategic level by the CNRH which is in charge of approving national water policies and general River Basin Management Plans. However, operational tasks of IWRM implementation are carried out by the ANA and may be delegated by the ANA to RBOs. The ANA itself takes the form of a water ministry since its director has the function of a minister. Several operational functions, formerly executed by sectoral ministries, have been dedicated to the ANA through the new law. If the ANA delegates some of its tasks to RBOs, further horizontal interplay can be achieved through memberships of representatives of water-related ministries, mayors of the respective river basin and the ANA. Hence, the National Water Law places strong focus on political and membership linkages at the strategic level without considering functional linkages at the lower level. Furthermore, repartition of roles and mandates of the river basin and local level remain unclear as these are to be assigned by the ANA. In addition, the processes of river basin management plan development, the formation of RBOs and River Basin Committees (RBC), the involvement of stakeholders and public participation is not described in the new law. According to the new law, local and regional governments take part in both RBOs and RBCs, but it is not clear what role they will play within their political borders, in particular concerning water regulation and management competencies (Novo and Garrido, 2010). The role of municipal governments has to be clear since Law 40 and Law 28 grant municipal and regional autonomy for regulating and managing water resources within their respective boundaries. However, the law assigns responsibility for granting authorizations for agricultural water use for small irrigation systems for lands smaller than 3 hectares. Concessions and licenses for large water and sanitation systems as well as for hydroelectric and agricultural purposes (areas of 20 hectares or more) are granted by ANA, while "for agricultural lands between 3 and 20 hectares, neither the water law nor its regulation states which agency is in charge of granting water use permits" (Novo and Garrido, 2010).

Apart from the general process on the national level, the IWRM process is also characterized as a multi-level process. This means that an integration of different policy levels from local to national level (or even to an international level if transboundary river basins are considered) is required. This should occur from top-down as well as from bottom-up. Municipalities have a long tradition of being in charge of managing the natural resources within their administrative boundaries and the elaboration of Environmental Management Plans (Plan de Gestión Ambiental) is common, but these policies are often developed without coordination with other organizations (e.g. neighboring municipalities or federal agencies) and are generally based on command and control measures or subsidies alone. Municipalities often do not have sufficient motivation for coordinated action at the river basin when applying regulative command and control or subsidy measures to improve natural resource use (Lee, 2000; Anderson et al., 2008). Hence, vertical interplay between the lowest policy level and higher policy levels is often lacking.

With regard to operational constraints, the present IWRM process still encounters insufficient financing, while roles and mandates between different governmental levels remain unclear. Insufficient financing is a clear result of the lack of implementation of financing instruments as defined in the water law. The law introduces financing instruments such as water fees as well as the instrument of PHES but how these should work still needs to be defined in special laws. As long as these laws have not been passed, the ANA, RBOs and basin committees do not have financial autonomy. Moreover, the lack of these financial resources impedes the institutional and organizational development. According to López Nolasco and Jiménez Otárola (2008a), the implementation status of the water law and its proposed instruments for financing is operationally insufficient for the river basin management application with tangible results. However, the lack of finance is basically a political problem causing important legal and organizational structures to remain drafts, for instance the law on water fees or the creation of the National Water Fund (cf., López Nolasco and Jiménez Otárola, 2008a; Novo and Garrido, 2010).

In 2008, less than 2 % of the national household budget was assigned to ministries and organizations in the environmental sector. The lack of human resources is equally evident. The National Forestry Institute (INAFOR) for instance has around 310 employees for the whole country (about one person per 425 km^2) and this ratio is similar for the Environmental Ministry (MARENA). These figures illustrate the institutional weaknesses, which are predominant in the context of compliance with environmental instruments of direct control and financing mechanisms provided by the national legislative framework (López Nolasco and Jiménez Otárola, 2008a). Moreover, this funding gap subsequently translates into capacity and information gaps. Accordingly, Novo (2010) summarizes the principal challenges of IWRM operationalization as follows:

- Lack of budgets, political will, personal capacity, information, monitoring and communication networks
- Legal contradictions between the Civil Code and the National Water Law on water use rights
- Problems with land property titles
- Political and hydrological boundaries do not coincide
- Intermediary structures between central and local governments are missing
- Absence of organizations and plans with respect to river basins

Moreover, Novo (2010) criticizes that the National Water Law is not known to the general population which impedes public participation in operationalization. The lack of knowledge about environmental legislation and policies among the population is wide-spread, especially in rural areas (Novo and Garrido, 2010), and in many cases the local needs and problems up- and downstream remain unaddressed and local environmental policies are often planned without involving affected (often upstream) populations (Lee, 2000). According to a survey among a broad group of stakeholders of the IWRM process in Nicaragua, Novo and Garrido (2010) identified that "[...] the lack of information is recognized as the major constraint for the water law implementation. By lack of information academia, international organizations, and public organizations refer to the lack of a comprehensive information system for both surface and groundwater withdrawals and land and agricultural input use". Hence, the lack of comprehensive water information and inefficient coordination mechanisms "[...] makes planning, control, and enforcement of the water law difficult and opens up the

Remaining tasks toward IWRM operationalization according to stakeholders
Updating and enforcing the legal framework and policies applied to water resources at national level • Updating the National Water Resources Policy and approving the National Water Resources Plan • Developing plans and programs for basins, sub-basins and micro-basins • Establishing of national and watershed committees • Preparing and approving of the Law on Fees for Disposal and Use of Water Resources • Encouraging the establishment of incentives for conservation actions, protection and rational use of water, through the development and approval of a Special Law of Payments for Environmental Services • Increasing the budget allocated to various governmental institutions related to the area of water resources, so that there is greater investment in equipment, technology and capacity building of human capital • Developing action plans for micro-basins • Protecting and conserving of the sources, especially in the upper watersheds • Approving the Law on Territorial Planning with watershed management approach
Strengthening community organizational structures • Promoting and strengthening watershed committees and irrigation districts • Strengthening local committees for the prevention, mitigation and response to natural disasters
Strengthening knowledge management at national level • Promoting education and awareness campaigns on the use and conservation of water sources • Studies to identify critical areas with high levels of risk • Preparing an inventory of all existing water sources at national level, indicating their current state and their main environmental stressors
Developing actions to ensure the quality of water for different uses • Regulating and controlling the use and extraction of groundwater for profitable uses • Regulating the application of chemicals used to develop agro-industrial activities

Table 6.1: Priority tasks for integrated water resources management at the national level (GWP, 2013b)

space for strategic behaviour“ (Novo and Garrido, 2010). As a result of these operational constraints, Novo and Garrido (2010) conclude that an advancement of the water law implementation on the level of municipalities seems more feasible following a bottom-up approach rather than a top-down one.

Akhmouch (2012) in authoring a OECD study on IWRM implementation in Latin America, highlights similar aspects as remaining governance challenges in Nicaragua: the mismatch between hydrological and administrative boundaries (problem of fit, i.e. administrative gap), horizontal co-ordination across ministries (horizontal interplay), lack of local and regional governments' capacity to design and implement water policies (capacity gap as an operational constraint), allocation of water resources across uses (residential, industrial, agriculture) and limited citizen participation. The *National Stakeholder Consultations on Water* recently carried out by the GWP (2013b) in Nicaragua to monitor the IWRM process draws similar conclusions and summarizes remaining tasks in order to advance IWRM operationalization (see Table 6.1).

The IWRM process in Nicaragua, thus, may be best described as still in an initial stage with regard to actual implementation and operationalization. While the National Water Law is in principle a standard and modern water law, implementing its guidelines and enforcing its rules represents a significant challenge in the Nicaraguan context (Novo and Garrido, 2010). As pointed out, there are still many legal, organizational and institutional steps required until the IWRM process may start to *drip down* to actual operational levels of river basins and below. This process will probably take several years and requires continuous political will and donor support. Moreover, the IWRM process as designed by the National Water Law runs the risk to become a typical blueprint approach from top-down which has been largely criticized for failing to operationalize IWRM (see Sections 2.2 and 2.3).

Hence, it can be conclude that the critical elements of IWRM operationalization concerning institutional fit and interplay at the operational level are still not in place. As a result decentralization and context-specific operational solutions to locally and regionally perceived problems have not been achieved. The principal reasons for this represent, on the one hand, typical operational constraints characterized by a general lack of funding, capacity and information and, on the other hand, a lack of mechanisms in the general IWRM policy of Nicaragua to solve the problems of institutional fit and interplay in relation to specific local contexts. The latter refers to a missing bottom-up complement of IWRM operationalization. Figure 6.2 illustrates how the problems of fit and interplay are addressed in the present IWRM process of Nicaragua. At the moment, the operationally important part of this process, namely to find the right management scale (grey shaded area in Figure 6.2) is still not provided by the organizations and mechanisms in place.

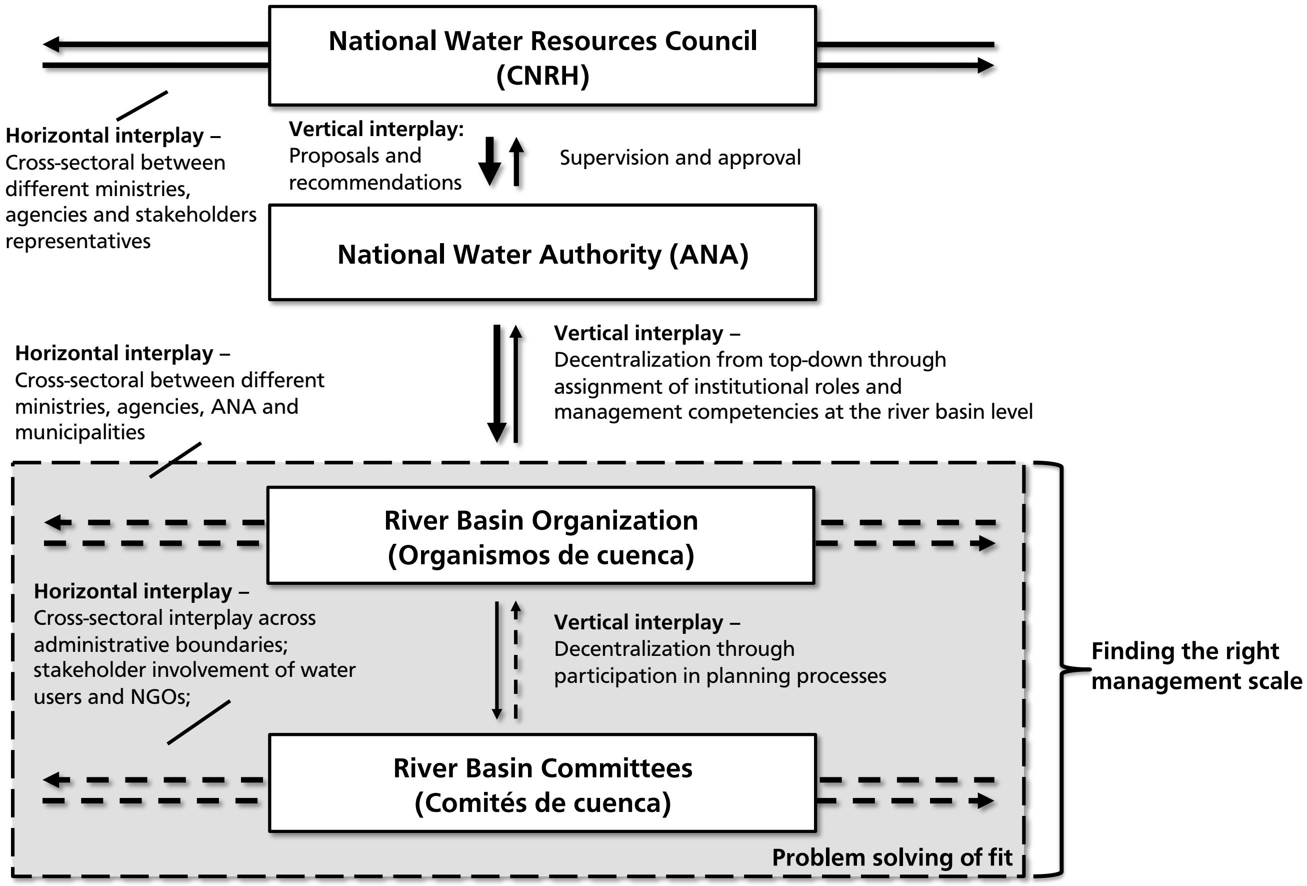

Figure 6.2: Institutional fit and interplay as addressed in the Nicaraguan IWRM process (Author's work)

However, the inclusion of PHES as a management instrument in the National Water Law raises hopes that this instrument may provide a complementary mechanism for context-specific IWRM operationalization from bottom-up. The law defines that the objective of PHES as a management instrument, is the elaboration of the economical, technical, legal and environmental basis required to organize a permanent payment. Hence, PHES schemes are considered to be a management instrument complementary to other economic as well as command and control instruments (Art. 14). In Art. 26, the law defines the realization of economic valuation studies and the analysis of water according to its form of supply, type of use and location as an ANA task in order to support the collection of water fees as well as payments for hydrological ecosystem services. Moreover, Chapter III of the water law is dedicated to hydrological ecosystem services. In Art. 93, the law states that the identification of hydrological ecosystem services should be given special attention in regions, basins, sub-basins and aquifers subject to advanced environmental degradation, with an increased risk of resource degradation, rapid changes in vegetation cover and wildlife extinction as well as risks to the population caused by climate change (ANA, 2010). Furthermore, the chapter specifies areas where PHES schemes can be established:

- Groundwater recharge, including secondary forests and tropical forests
- Springs
- Contaminated receiving water bodies
- Over exploited aquifers
- Wetlands
- Natural and artificial reservoirs and estuaries
- Lakes, lagoons, rivers of touristic, recreational or productive use with quantitative and qualitative problems (Art.94)

Finally, in Art. 95 the ANA is declared responsible for the implementation mechanisms of claims and payments for hydrological ecosystem services in order to finance PHES schemes in a sustainable way. To fulfill this task, the ANA will ask other organizations for support and participation. At the end of Chapter III of the law, the nature of PHES is clarified as being an incentive for the conservation, protection, and rational use of water and other natural resources of determined river basins. PES will be regulated through a special law. The ANA will supervise that suppliers of hydrological ecosystem services will receive a fair compensation and payment for the services they supply.

Apart from recognizing PHES as a management instrument in the National water Law, the PHES instrument has been referred to repeatedly in several other laws and decrees. An overview of environmental laws concerning IWRM is given in Table 6.2.

Name of Law	Reference to PHES	Year
Law No. 40: Law of Municipalities (Ley de Municipios)	Assignment of municipal sovereignty (decentralization) of environmental policies and financing; Municipal competencies include the development, conservation and control of a rational use of the environment and natural resources as a basis for sustainable development of the municipality and the country by promoting local initiatives and contributing to monitoring, surveillance and control of protected areas in coordination with federal agencies	1988
Law No. 217: General Law on the Environment and Natural Resources (Ley General del Medio Ambiente y los Recursos Naturales)	Creation of the National Environmental Fund (Fondo Nacional del Ambiente), which aims at the development and financing of programs to improve environmental protection, conservation, restoration and sustainable agricultural production. Contributions to this fund ought to come from environmental licensing (e.g. for timber extraction), environmental fines and taxes, donations and additional sources	1996
Law No. 462: Law of Conservation, Formation and Sustainable Development of the Forestry Sector (Ley de Conservación, Fomento y Desarrollo Sostenible del Sector Forestal)	Creation of the National Forest Development Fund (Fondo Nacional de Desarrollo Forestal) as an autonomous funding source to finance strategic programs and projects for the forestry sector. At present, the fund prioritizes the financing of actions of social interest, reforestation and restoration as well as the protection of source areas for domestic water supply in river basins. PES are introduced as one option for financing. Three PHES projects are financially supported by this fund (FONADEFO, 2012)	2003
Law No. 475: Law on Citizen Partizipation (Ley de Participación Ciudadana)	Guarantees the right to form associations of citizens and organizations of different sectors to give all members of society the opportunity to participate in local instances for the formulation of public policies, self-organized development projects and to realize programs in order to improve the provision of public community services. By municipal decrees (see Law of Municipalities), local management committees for PHES or water management in general can be assigned certain tasks	2004
Law No. 647: Law on Reforms and Additions to Law No. 217 (Ley de Reformas y Adiciones a la ley No. 217)	Inclusion of PES as an environmental management instrument within the General Law on the Environment and Natural Resources (Law No. 217). Creation of a system of valuation and Payment for Ecosystem Services as an environmental management instrument with the objective to value and establish PES and to generate financing and incentives for environmental conservation, preservation and sustainable use of natural resources	2008
Law No. 559: Special law on Delinquencies against the Environment and Natural Resources (Ley Especial de Delitos Contra el Medio Ambiente y Recursos Naturales)	Regulation of violations of the General Law on the Environment and Natural Resources (Law No. 217)	2005
Decree: 01-2007: Regulation of Protected Areas (Reglamento de Áreas Protegidas)	Regulation of Protected Areas	2007
Law No. 620: National Water Law (Ley General de Aguas Nacionales)	Conceptualization of Payments for Hydrological Ecosystem Services. Introduction of water fees for the use and diversion of water	2007

Table 6.2: Nicaraguan environmental laws and references to the Concept of PHES (Author's work)

Although the PHES instrument has received considerable recognition in the Nicaraguan environmental legislation, a national PHES scheme, as in the case of Costa Rica or Mexico, has not been established. Nevertheless, the Nicaraguan approach pursues the establishment of national funds (for hydrological or general ecosystem services) again designating a central role of the ANA, in the case of hydrological ecosystem services, as intermediary for the implementation of schemes. However, so far not even one PHES has been implemented by a central governmental institute or agency such as the ANA. By contrast, several locally organized and user-financed schemes have been implemented in the last decade. River basin management in Nicaragua is largely concerned with managing land for the right use in order to prevent negative impacts to downstream water uses. Nicaragua is an agriculturally dominated country with a very small industry sector and its main water users are public water suppliers, public hydro power producers and mostly private irrigated agriculture. All of these water users rely on (upstream) areas for water provision. The PHES concept is addressing these circumstances directly and is gaining popularity in Nicaragua for this reason.

PHES may possibly be more than just a means of establishing economic incentives for conservation, but also a means for the operationalization of IWRM contributing to address problems of spatial fit and institutional interplay more directly and context-specific. This additional objective of PHES may justify higher transaction costs and efficiency trade-offs since these transactions are also required as part of a necessary IWRM process.

6.2 Application of the PHES instrument in Nicaragua

While the formal IWRM process with the creation of an enabling environment by laws and national policies, the establishment of institutions and the assignment of responsibilities as well as funding is slowly progressing from top-down, the implementation of Payments for Hydrological Ecosystem Services schemes at the river basin level, in form of locally organized schemes, is several years ahead. In 2000 the first Nicaraguan PHES schemes entered a preparatory phase with bio-physical assessments of the state of natural resources (land use assessment, soil mapping and water resources assessment) at the respective river basin level. At the same time, different stakeholder groups (water and land users as beneficiaries and providers of hydrological ecosystem services) were identified and economic valuations of the hydrological ecosystem services were carried out as part of this preparatory PHES process. 11 locally financed payment schemes (three of them were implemented as environmental funds) for hydrological and other ecosystem services, mainly biodiversity and carbon sequestration, have been documented until now (Talavera, 2007; Kammerbauer et al., 2010). Examples of payment schemes considering hydrological ecosystem services are summarized in Table 6.3.

Municipalities	Sub-basin; size	Service buyer and immediate beneficiaries	Start
Belén, Buenos Aires and Potosí	Río Gíl González; 68 km^2	Private sugar company CASUR, plantain producers, public water supplier for Belén, Buenos Aires and Potosí	2008
Río Blanco	Río La Golondrina; 10 km^2	Municipal water supplier (Empresa Municipal Aguadora de Río Blanco; EMARB)	2002
El Regadío, Tisey and La Estanzuela	Río Estelí; 27 km^2	Municipal water supplier	2007
San Pedro del Norte	Río Paso los Caballos; 72 km^2	Municipal water supplier	2001

Table 6.3: Case study examples of Payments for Hydrological Ecosystem Services schemes in Nicaragua (Author's elaboration)

Almost all PHES schemes have been initiated with the support of international development cooperation agencies as intermediaries in order to promote financial sustainability of environmental conservation actions. As illustrated in Table 6.3, several laws in Nicaragua, issued after the PHES had commenced, include some references to PHES, but are still far from being implemented or institutionalized. Thus, the establishment of legal and institutional frameworks on the national level seems to lag behind what has already been established, in practice, on the regional or local level.

The following section documents the implementation of the PHES scheme in the Gíl González river basin in Nicaragua as a case study example. The Gíl González PHES scheme is deemed a successful locally organized scheme and was selected as a case study for The Economics of Ecosystems and Biodiversity (TEEB) initiative this year (Hack et al., 2013). The author was involved as an external expert during the whole implementation phase of the scheme from 2008 until 2011 and the subsequent and still ongoing execution phase. The following documentation of the implementation process illustrates how practical PHES implementation provides potential opportunities to solve common problems of spatial fit and institutional interplay as described in Sections 3.1 and 3.2. Furthermore, it is the basis to assess the instruments functional role within a broader policy mix for IWRM implementation.

6.2.1 The payment for hydrological ecosystem services project in the Gíl González river basin

The Gíl González river (sub-)basin is located in the southwest of Nicaragua forming part of the Rio San Juan river basin, the largest river basin in Nicaragua. The Gíl González river flows from west to east with a length of about 25 km until it empties, after passing through the Ñocarime Lagoon, into the Lake of Nicaragua, the greatest freshwater reservoir of Central America. The Gíl González river basin is shared by the municipality of Belén (upper part of the basin) Potosí and Buenos Aires (lower part of the basin).

With about 67 inhabitants per km^2 the river basin has a moderately high population density (INIFOM, 2011). While the lower part of the river basin is more densely populated and agricultural production is more intensive, the upper part of the basin is characterized by dispersed settlements, few infrastructural developments and predominantly subsistence agriculture of poor farmers (Hack et al., 2013).

The climate of the region is tropical with a prolonged dry season of five to eight months, with average annual precipitation between 1000 and 2000 millimeters and a generally bi-modal pattern of rainfall, with a shorter and a longer dry period. Given that the prevailing winds in the ecoregion blow from the northeast or east to the southwest or south and most of the ecoregion has mountain systems running from northwest to southeast, the Pacific side of Central America, where the Gíl González river basin is located, receives a lesser amount of rain than the Caribbean side. Typically, dry sub-tropical forests can be grown on a wide variety of soils and represent the original endemic vegetation in lowland and pre-montane areas from zero to 800 meters elevation (Marín et al., 2005; González-Rivas, 2005; Tarrasón et al., 2009; Dirzo et al., 2011).

However, in the last decades large parts of the upper river basin have been deforested due to agricultural expansion for the cultivation of corn, beans and rice on a subsistence level and the need for wood for construction as well as for cooking. Deforestation is a serious problem for the whole region as it accelerates soil erosion, decreases agricultural production and increases turbidity, which directly affects downstream water users (Dirzo et al., 2011). Importantly, it is assumed that deforestation decreases the amount of recharge to aquifers by increasing surface runoff. Deforestation along rivers also increases the risk of the rivers drying up because of the increased exposure to the sun.

Today, extensive cattle ranging is often practiced as well, as a means to diversify farm income. However, the more fertile lowland areas are cultivated by richer farmers who cultivate sugarcane, rice and plantains. This translates into a situation where the more disadvantaged part of the population is forced to cultivate in the upper areas of the river basin due to higher land prices in the more fertile and accessible lowlands (Hack et al., 2013). Thus, fragile soils and ecosystems of the upper river basin areas are damaged resulting in a reduction of infiltration rates and therefore a reduction of river discharge and lowering of the groundwater table, soil erosion, and contamination of the river system by animal feces (Hack et al., 2013). Hence, any change of poor farmers' existing production systems to more sustainable systems would result in additional costs for a largely marginalized population group.

Empirical data from the Gíl González river basin, based on hydrological assessments of individual watershed areas with different land uses and measurements of water tables of local wells (CIRA, 2007; Hack, 2010, 2011), indicates that the complex land use change of the past has led to decreased river discharge, especially during the dry season from November until April, and to a deterioration of the river's water quality. Apart from ecological problems, this has also resulted in economic consequences for the downstream water users, particularly to the privately owned sugar company CASUR. The company cultivates sugarcane on 54 km^2 along the shore of the Lake of Nicaragua from which it produces sugar, molasses and energy. It is the biggest water user of the region. 10 km^2 of the total cultivation area are located within the lower part of the Gíl González river basin. About half of the sugar production is exported (reference year of 2008). With 1600 direct and several thousand indirect employees the company is also an important employer for the region.

The sugarcane production is strongly dependent on the availability of surface and ground water for irrigation during the dry season. The increasing scarcity of inland water resources, i.e. decreasing river flow as well as lowering groundwater tables, and deteriorating water quality (CIRA, 2007) convinced the company to invest in the upper part of the Gíl González river basin to improve the hydrological service provision. This investment resulted in the establishment of a payment for hydrological ecosystem services concept as a Public-Private-Partnership (PPP) project between the German Agency for International Cooperation (GIZ), the municipality administration of Belén and the private sugar company CASUR in Nicaragua.

The implementation process of the PHES in the Gíl González river basin is very similar to the typical implementation process of local PHES as analyzed in Section 5.2.1. Hence, a distinction can be made between a preparatory as well as an application and execution phase.

The preparatory phase includes the identification of principal bio-physical cause-effect relationships between land use and service provision. The scheme in the Gíl González catchment considered the provisioning and regulating services of freshwater in terms of water quality, quantity and especially timing of flow. In this context, especially the role of shrub vegetation, secondary forests and dry tropical forest cover for water infiltration into the ground and as a regulator for torrential precipitations in the rainy season was investigated.

The PHES scheme in the Gíl González river basin had its origin in the revision of the municipal strategic development plan of the municipality of Belén and a redirection toward a development plan with focus on river basins incorporating elements of participatory land use planning and watershed management. The municipality recognized the severe environmental degradation within its territory and decided to apply the PHES instrument to reverse this degradation. As one methodological step of the plan preparation, the existing river systems and watershed areas were prioritized regarding their importance as drinking water for the rural and urban population and for the agricultural production of the former.

The territory of the municipality of Belén forms part of seven river basins. The prioritization of a river basin was based on 14 criteria: basin surface within the municipality's territory, population density of the river basin, existence of prior studies of water quality and quantity, existence of areas prone to flooding, water contamination, use conflicts, protection zones (actual and potential), irrigation water use, agricultural use potential, forestry use potential, social organization and productivity, status of access roads and presence of supporting NGOs. Moreover, the river's ecological function regarding the Nicaragua Lake was also taken into consideration.

The decisive criteria for the prioritization of the Gíl González river basin were the largest surface within the territory of the municipality, the second highest population density, the existence of a prior study on water quality and quantity, the level of social organization and productivity as well as good accessibility. Moreover, the Gíl González river basin was chosen for being the most important mainly due to the fact that the urban population of Belén derives its drinking water supply from this river system. Furthermore, the Gíl González river basin provides agricultural plantations, sugar cane and rice producers with irrigation water.

In this second step, using GIS technology, satellite imagery from 2005 was interpreted to determine land use in the municipal territory on a scale of 1:50,000 (INTELSIG, 2008). Additional information on land uses in the recent three decades was retrieved from aerial photographs of the project site. This land cover information was overlapped with soil type information, hang slopes and geological features in order to determine land use conflicts. Information on soil classes and land use was provided by recent field surveys (Acuña et al., 2008a,b,c; Rodríguez et al., 2003a,b). Further hydrological data was gathered from 6 rain gauges with daily records within a range of less than 20 km from the catchment and climatological data of 3 stations with daily records of temperature, wind speed, humidity, air pressure, insolation with a distance of less than 30 km from the catchment. Additionally, there are measurements of river discharge and groundwater tables from recent years (CIRA, 2007; Hack, 2011). A summary of the information available prior to the PHES scheme execution and its temporal and spatial resolution is given in Table 6.4.

Data	Temporal resolution	Spatial resolution
Precipitation	1958 - 2008 (daily)	< 20 km from the catchment
Climate	1999 - 2007 (daily)	< 30 km from the catchment
River discharge	1965, 2006, 2008 - 2009 (monthly)	Every 5 km along main channel
Soil	2002, 2006, 2008	Whole catchment field survey
Vegetation and land use	2004, 2008	Whole catchment field survey
Hydrogeology	2006 (single)	3 wells in the catchment
Potable water production	2003 - 2008	Lower catchment
Census	2005	Village level

Table 6.4: Geophysical and socio-economic data base (Hack, 2010)

These materials were used to identify and classify hydrologically critical areas of the river basin. The source areas of the headwaters as well as riparian areas of the main stream and its tributaries were prioritized for protection. Within the Gíl González river basin, also areas of moderate and severe land use conflicts were determined which would be the areas where conservation, land cover recuperation and soil conservation measures should be implemented in order to promote future rainfall infiltration rates.

Parallel to this process the Center for Aquatic Resources of the National Autonomous University of Nicaragua (CIRA-UNAN) was involved in establishing a baseline for the monitoring of river discharge volume and water quality (CIRA, 2007). The initial bio-physical assessment was mainly driven by intermediaries acting as facilitators for the development of a PHES scheme. Besides the GIZ as donor and project initiator, the Association of Municipalities of the Rivas Department (AMUR) and the Center for Aquatic Resources contributed with assessments of soils, land uses and water resources too. The Association of Municipalities of the Rivas Department is an organization represented by the 10 governments of the member municipalities of non-profit and non-partisan character, whose main objective is to strengthen municipal autonomy. A principal objective of AMUR is the development of land use planning in the Rivas Department.

The bio-physical assessment carried out identified severe land use changes in the past as the principal cause for the degradation of hydrological ecosystem services (CIRA, 2007; Acuña et al., 2008a,b,c). In a second step of the preparatory phase, the principal beneficiaries of these services were addressed as part of a socio-economic valuation process. The private sugarcane producer CASUR was identified as the main beneficiary, while the municipal water supply of Belén provided by ENACAL and commercial producers of plantains and vegetables were determined as benefiting as well. All beneficiaries are characterized by extractive uses of water. The most critical time of service provision is at the end of the six months long dry season from February to April. Besides quantitative aspects of service provision, benefits are derived from improved water quality, especially in the case of the municipal water supply, where high water quality standards are demanded.

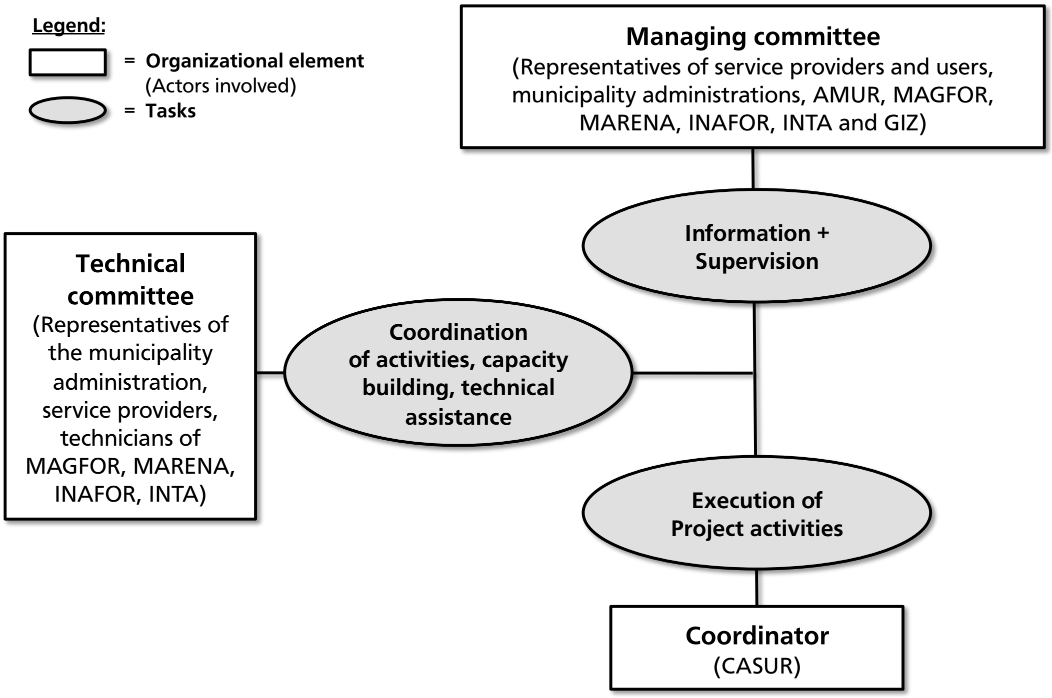

Figure 6.3: Organizational framework supporting the payment scheme; based on (Hack, 2010)

At first, the GIZ contacted CASUR to obtain information on water consumption and willingness to pay for service provision (AMUR, 2013). The municipality of Belén joined this socio-economic valuation process as service providers are located within its municipality. The previously identified areas in the Gíl González river basin with land use conflicts, then allowed identifying potential hydrological service providers. Within the selected local communities a broader participatory land use zoning process was initiated and sustainable use standards were agreed upon. In order to identify opportunity costs of land use changes, it was considered how much had to be paid to recuperate degraded lands in the upper part of the catchment area and how much had to be paid to protect intact dry forest remnants and gallery forests. Incentives to introduce soil and forest, i.e. land use conservation measures were negotiated, in cooperation with the local population. Finally, the participants of the valuation process decided to orientate the payment amount on the opportunity cost each farmer would incur if he left land out of production, i.e. a hypothetical annual average rent per 1 ha of his land. A study executed in Belén to determine land prices concluded that the medium land rental price per ha in the middle and upper watershed is US-$ 36 / ha / year (Hack et al., 2013). Thus, the valuation and negotiating process in this case study was focused on the opportunity cost of land use changes representing the minimum payment to achieve service providers' willingness to accept. The willingness to pay, on the beneficiary side, was not explicitly assessed. However, the main beneficiary, CASUR, accepted the determined payment amount, regardless of actual benefits of service provision being higher or lower. This procedure of input-based service valuation is typical for local PHES, as documented in different case study reviews (see Section 5.2).

Finally, the concept of a payment for hydrological ecosystem services scheme was introduced with the support of the local municipality, the GIZ and CASUR in form of a public private partnership (Flores Barboza et al., 2011; AMUR, 2013). A cooperation contract was signed by the three supporting parties, establishing individual duties of each party in order to implement the PHES scheme. An executive office for the scheme administration was installed at CASUR. Furthermore, a scheme coordinator was contracted and a managing committee was constituted as the body responsible for the management of the PHES scheme. The managing committee consists of representatives of the municipality of Belén, CASUR, the GIZ, AMUR, governmental ministry branches of the Rivas Department (MAGFOR, MARENA, National Forestry Institute INAFOR, National Agricultural Institute INTA) and representatives of hydrological ecosystem service providers. Beside the managing committee, there is also a technical committee which is composed of the scheme coordinator, technical personnel of public authorities (municipality, INTA, INAFOR, MARENA, MAGFOR), and representatives of service providers. The technical committee is in charge of elaborating and implementing capacity building activities and provides general technical assistance. Figure 6.3 illustrates the scheme's organizational framework, the composition of all parts of the framework and their respective responsibilities.

The organizational framework regulates and supervises the transactions between service provider and service buyer allowing the realization of payments. To support a participative decision making regarding the design of the payment

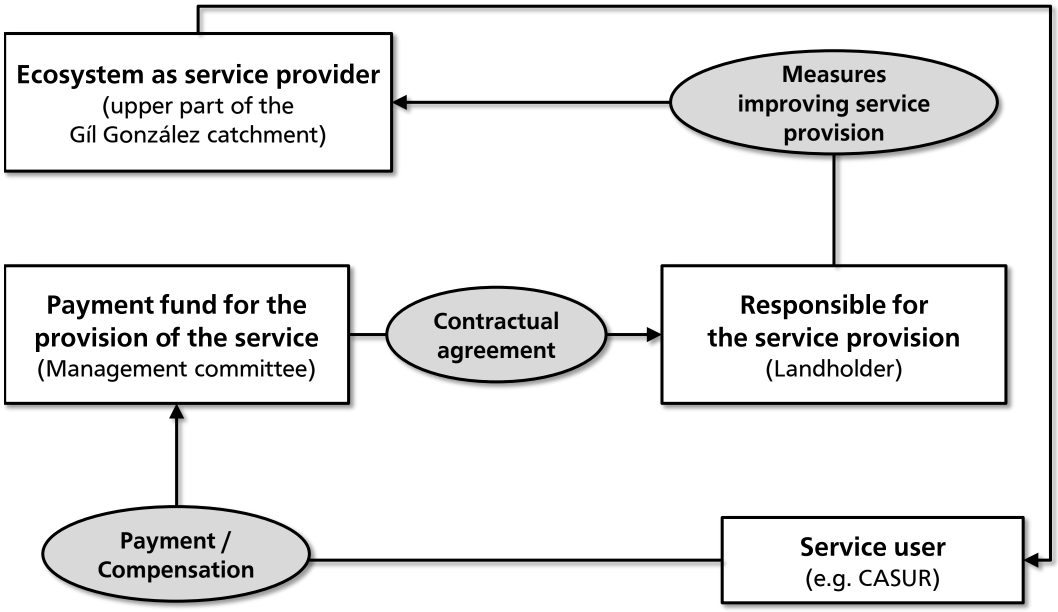

Figure 6.4: Payment scheme of the Gíl González catchment; based on (Hack, 2010)

scheme and the definition of measures to improve the service provision, all stakeholders are equally represented within the organizational framework. Therefore, the established organizational framework, consisting of a managing committee, a technical committee, and a coordination unit, includes representatives of the service providers (land users of the upper river basin), the service buyer, the local municipality administration, the GIZ and technicians from several federal ministries for capacity building and technical assistance concerning conservation measures.

The payment scheme is supervised and monitored by the multi-stakeholder management committee. Thus, payments are not executed directly, but intermediate through a fund which is administered by the managing committee. With the engagement of service providers and buyers as part of the management committee, a greater ownership, transparency between different actors and the general public as well, increased trust among the stakeholders could be achieved (Flores Barboza et al., 2011).

The service buyers pay a fixed amount per year for the use and conservation of the hydrological service into a project fund and service providers receive an annual payment for taking their formerly cultivated land out of production and instead reforest and conserve existing forest. Most of the land now under protection or reforestation was previously used either as pasture for cattle or for the cultivation of maize, beans or rice. In one case forested land was included as well in order to protect it. The payment scheme is shown in Figure 6.4.

Measures	Description; payment / compensation
Conservation of existing forest	Protection of exiting forests; 36 US-$/ha/year
Expansion of existing forest	Reforestation of degraded areas; 36 US-$/ha/year
Expansion / recuperation of gallery forest	In a period of 4 years every farmer had to plant annually at least 300 trees in a 50 meter stretch from the river bed until a 200 meter broad stretch was reached (established by the Environmental Law to protect forests along rivers); 36 US-$/ha/year
Establishment of complex fences	The participating farmer had to remove the vegetative material and establish the living fences. The National Forest fund paid an additional 0,5 US-$ for each tree established. In addition every farmer received: a) barbed wire in order to protect forest from cattle grazing and b) different types of endemic fruit tree seedlings
Additional measures required to receive payment	Establishment of fire protection lines around forests (*Rondas*). Removal of dead wood material in forests in order to avoid forest fires. Participation of land users in technical workshops with topics demanded by farmers. Precondition to receive payment for the above mentioned measures.

Table 6.5: Summary of conservation measures and respective payments or compensations; based on Hack et al. (2013)

In order to participate in the PHES scheme, every service provider had to sign a formal contract, in which the mutual agreements and obligations were formally established. Moreover, participating land users only received payments after a technician of the technical project committee had visited and evaluated the accomplishment of the agreements (AMUR, 2013). Hence, the PHES scheme contains conditionality on input, i.e. compliance with agreed conservation measures and (monetary) payments are realized ex-post.

Table 6.5 shows the list of conservation measures for which farmers received remuneration in cash and remuneration in kind.

The introduction of the PHES schemes resulted in a very high initial demand to participate on the service provider side. However due to a limited initial financial volume of the fund, the project had to limit the number of signed contracts to those land users who still had forested areas on their lands, and focused on those who were working in areas which required an urgent regeneration of vegetation cover. At the beginning, only the municipality of Belén and CASUR, as service buyers, and the GIZ contributed to the payment fund. In 2010 and 2011, the National Fund for Forest Development (Fondo Nacional de Desarrollo Forestal, FONADEFO) and INAFOR with funding provided by the National Forest Program (NFP) of the FAO, also contributed to the fund (see Table 6.6).

Payer type	Name	2008 and 2009	2010 and 2011	2012	Total
Service buyer	CASUR	48,000	32,000	33,140	111,140
	Municipality of Belén	12,000	9,741	7,826	29,567
					140,707
Governmental funds	FONADEFO	-	20,000	-	20,000
	INAFOR (FAO-Facility)	-	10,500	-	10,500
					30,500
International donor agency	GIZ	52,000	35,000	8,175	95,176

Table 6.6: Financial contributions of water users and donor agency (US-$); based on Hack et al. (2013)

When the payment scheme was implemented, interested land users had to present registered land titles (legal property rights) since a legal agreement had to be signed between service buyer and service provider to ensure mutual commitment. This circumstance limited the participation of land users as service providers to a certain extent and only 105 ha of land with official land titles could be included in the scheme for improvements in hydrological service provision. This area was considered far too small to have a considerable impact on water production. Therefore, the project decided to apply a more flexible model in which farmers who presented a formal statement from the cadastral department of the municipality that no land claims from third parties existed, were able to take part. Since 2008 the number of participating farmers has increased continuously.

6.2.2 Impact and success factors of the PHES scheme

The PHES scheme was an important means to initiate a dialog among different public and private actors at local and regional levels such as up- and downstream land and water users, people in rural and urbanized areas and different local governments. Before the scheme's establishment, no open dialogue existed between the upstream municipality of Belén and the downstream hydrological service demanding sugar cane producer. However, mental blockades on both sides had to be dismantled in order to create a constructive negotiation basis. On the one hand, the sugarcane company CASUR had to struggle with a negative image regarding their environmental impact due to the practice of burning harvest residues and cutting trees in order to fuel their sugar refining process. On the other hand, the municipality of Belén was forced to act against advancing environmental degradation in the upstream part of its territory. In this situation, the hydrological ecosystem service concept seemed to appeal to both side as a feasible way to finding a solution. The bio-physical assessments, which were carried out in the preparatory phase of the scheme, provided scientific and technical arguments that convinced CASUR of the need for hydrological service provision (AMUR, 2013). Hence, a main factor for the successful establishment of the PHES scheme was CASUR's demand for these services. CASUR recognized its dependence on hydrological ecosystem services and the resulting costs due to their deterioration. As a result, the company was willing to invest in the protection and the restoration of parts of the ecosystem. In addition, CASUR gained a positive recognition from society through the initiative, which was also an incentive to participate in the scheme.

Moreover, land users in the upper part of the river basin were willing to change their land use in order to comply with the service provision requirements. A certain awareness of the importance to act by mitigating the present environmental problem, that had originally been caused by their own collective action, encouraged their participation, although the payment does not fully compensate their opportunity costs.

Having discovered common interests among these critical stakeholders, a continued moderated dialog between the stakeholders finally made an institutional arrangement possible. As in many other locally organized PHES schemes, this initial moderating process was supported by an intermediary, in this case the GIZ.

Besides its role as the principal service buyer, CASUR acted as an important facilitator for the execution of the scheme by providing accommodation for the scheme coordinator and also employing the coordinator. Moreover, CASUR carried out several discharge measurements for baseline identification of service provision and provided further climatological data from the company owned climate station.

The conformation of the managing committee, including the representatives of the municipality of Belén, CASUR, the GIZ as well as upstream land users as service providers was very important for the acceptance by potential service providers as well as for trust building. Through the involvement of representatives of service providers in the managing committee, they could participate with voice and vote in the decision-making process concerning the scheme design, e.g. with regard to the interpretation of conditionality and additionality. Moreover, the service providers have the ability to express their views and concerns about the scheme's progress.

The management committee for the river basin was constituted and now plays an important role in the development and implementation of sustainable land management plans that consider protection and cultivation zones. Also, the committee structure guarantees the selection of participating land users based on scientific criteria, e.g. land users possess land in areas formerly identified as hydrologically sensitive and are willing to apply appropriate conservation measures, avoiding the introduction of political elements and allowing for a transparent monitoring of provider's compliance with the signed contracts since all participating stakeholder are equally represented. With the establishment of an organizational framework consisting of representatives of all stakeholder groups, development and management of water resources is starting to get more participatory, following a cross-sectoral approach.

Furthermore, due to the increased public awareness of environmental problems and the success of direct payments to service providers, land users now participate much more actively in the municipal council meetings in which the municipal budgetary lines for next year's investments are agreed upon (Hack et al., 2013). In these meetings, land users now defend the financial contribution of the municipality to the PHES fund and even demand the increase of financial contributions to this fund. The PHES process changed people's perceptions on water, which is now considered as a finite and vulnerable resource while gaining economic importance. Participants as well as non-participants of the payment scheme recognize environmental protection as a main objective of the payment concept. These changed perceptions were expressed during a field survey (Schuchmann, 2010) and at several workshops which formed part of the systematization process of this project experience (Flores Barboza et al., 2011). In general, there is a lot of interest to participate in the project as a service provider. Problems with access to the payment scheme because of uncertain property titles and possible marginalization of non-participants were not expressed by the interviewees (Schuchmann, 2010).

Through the PHES scheme, an area of 513 ha is now (2013) under conservation and 36 km of complex fences (wind breakers) have been established (see Figure 6.5). Furthermore, a total of 93 hydrological ecosystem service providers have been engaged in these actions and have been compensated either in kind or through monetary payments (AMUR, 2013). The initiative contributed to the conservation of an important area of the upper part of the river basin ecosystem and public awareness regarding the costs of ecosystem degradation and the interconnection of ecosystems in the uplands and lowlands was raised considerably (Hack et al., 2013).

Figure 6.5: Images of conservation zones established within the PHES scheme (Source: Author)

	2008	2009	2010	2011	2012	Total
No. of contracts	29	-1	+10	+60	0	98
No. of service providers	29	-1	+10	+55	0	93
Male service providers	21	-1	+8	+45	0	73
Female service providers	8	0	+2	+10	0	20
Hectares contracted	105	+3	+72	+332	-5	507
Km of complex fences	0	0	+36	0	+3	39

Table 6.7: Evolution of project participants and conservation measures from 2008 - 2012; based on Hack et al. (2013)

However, the actual improvement of service provision still remains uncertain since longer monitoring is necessary to be sure about the conservation effects. Another important challenge is the introduction of conservation measures to improve water quality, since activities have been addressed primarily to water quantity until now. Although the conservation measures applied to improve the available water quantity, may also have a positive impact on water quality, e.g. reduced soil erosion and agro-chemicals, some sources of water quality deterioration have not yet been addressed, e.g. problems with coliform bacteria introduced by animal feces. Finally, a better understanding of the relation between land uses and their impacts on hydrological services would contribute to the implementation of more effective conservation measures.

The introduced payment scheme for hydrological ecosystem services is very perceived positively by the participating land users as a secure and continuous income source over the negotiated contract period of five years, although some beneficiaries regard the project solely as an environmental protection measure without significant economic importance to them. In addition, land users value future benefits from reforestation such as fruits, fuel wood and an increase of land value. About half of the beneficiaries would include more land into the payment scheme and there is a high interest to participate in the scheme as a service provider (Schuchmann, 2010), although the amount of payments does not fully cover the economic losses if land is completely taken out of production for reforestation. At the same time, the maintenance of reforested areas is labor intensive and does not imply additional reward. Therefore, it is necessary to diversify conservation and restoration measures to generate additional income. In 2012, at least the payment amount per ha and year could be raised from 36 US-$ to 41 US-$ and additional in-kind payments (e.g. material and tools) could be provided. Table 6.7 illustrates how the number of participants and the size of the conservation area have increased since 2008 (first year of payments).

When considering the principal issues of potential ecosystem service providers' enrollment, compliance of ecosystem service provision and additionality as well as land use-ecosystem service provision linkage, required for the proper function of PHES (as defined by Wunder and Albán (2008) and discussed in Section 5.2.3), the case of the Gíl González PHES could solve initial problems of provider enrollment and achieved expected levels of compliance with agreed conservation measures. Furthermore, there is sufficient reason to believe that additionality could be achieved. However, whether the measures taken will in the end affect hydrological service provision is still unclear since the evaluation and monitoring period is still too short. The monitoring and measurement of river discharge and groundwater levels initiated during the implementation of the scheme has to be continued to verify assumed land use linkages to service provision.

Effects of leakage and perverse incentives resulting from the implementation of the PHES scheme have not yet been documented. The question of permanence of the scheme has to be considered differentiated. Several measures introduced by the scheme do not require permanence in terms of payments, for instance the establishment of complex fences as wind breakers and the protection of riparian areas, since they are based on voluntary agreements. However, continuous payments are negotiated for reforestation and the protection of forested areas. The municipality of Belén and CASUR continue to contribute with financial and other resources without a designated end. Negotiations with additional service beneficiaries such as ENACAL and the cooperative of plantain producers (APLARI) have been initiated but have not achieved agreements. Their ability to pay may differ from that of the sugar company and has to be considered along with possible effects on the end users of their products. In case of the water supply company the local water charge may increase (financial overloading of poor water users / service buyers since water has to be, to a certain extent, considered as a basic good). The integration of additional water users in the project holds the possibility for conflict resolution, because former competitive water users could share common interest as service buyers and this may open room for additional bargains and water saving strategies. In order to identify their water use (water source and volume) and plantain production, 25 % of the registered plantain producers of the municipalities of Belén, Potosí and Buenos Aires have been interviewed. Based on this information a payment per area of cultivation was proposed to the plantain producers.

On the supply side, the option of service provision has to become a serious alternative to unsustainable (agricultural) land use by increasing its economic importance. A more feasible way could be to look for synergies between certain agricultural (e.g. agroforestry) and non-agricultural (e.g. rural tourism) practices and the provision of hydrological services so that an income complementary to the payment can be achieved. In this context it is also important to find a way to generate complementary income in the short term and to take non-monetary benefits (i.e. compensations) into account. These synergies could also represent a viable strategy to establish alternative or additional development perspectives for rural communities and reduce the risk of creating perverse incentives, e.g. receiving payments for land use change that is already enforced by law.

6.3 Contributions to solve the problems of institutional fit and interplay

In the following section, the contribution of the different PHES implementation processes towards IWRM operationalization is assessed. The assessment illustrates the specific contribution to solving the problem of spatial fit, improving institutional interplay and improving local decision-making. In the context of water resources management integration, in the first place, refers to deliberately moving away from fragmented approaches of sector policies. As far as the physical integration within a river basin is concerned, this involves integration of land and water management, surface water and groundwater management, water quantity and quality, and upstream and downstream water-related interests. This challenge was introduced in Section 3.1 as the problem of spatial fit.

The implementation process of PHES schemes in Nicaragua, like the one in the Gíl González river basin, in general departs from a common problem situation which is based on a shared belief on land use and hydrological services linkage. It is thus commonly believed that changed land use in the past years or more often decades, e.g. deforestation for agricultural purposes or pasture, in upstream areas of the river basin, had caused rivers to run dry, groundwater tables to lower and/or water quality to decline. It is further believed that a reversal to former, closer to natural state land uses can enhance water quality and quantity downstream. Thus, the assumed physical cause and effect relationship is based on the geographic unit of the river basin. Actually, because of the size of many PHES schemes in practice, some river basins should be called sub-basin or micro-basin since the basin size in question varies from 10 km^2 to 70 km^2 (see Table 6.3). However, as the PHES example of the Gíl Gozález river basin shows, this spatial extent represents the actual level of local impact and problem perception. In the case of the Gíl González river basin, the positive correlation between forest coverage and dry season flow was perceived as a proven fact because in micro-basins where the portion of forest left was higher than in others of similar size, dry season flow was present while in others it was not (Flores Barboza et al., 2011). The logic of upstream lands that provide hydrological ecosystem services and downstream beneficiaries, which are connected through the surface or sub-surface water flow of a river basin, is inherent to the PHES concept. Therefore, it addresses water resources management from "source to tap" (cf. Mitchell, 2005). The policy instruments of Environmental Management Plans (Plan de Gestión Ambiental) and Local Development Plans (Plan de Desarrollo Local), which have been developed by the municipalities for their respective administrative boundaries, are now developed in the PHES scheme of the Gíl González river basin on the river basin basis, considering the administrative areas of the three municipalities Belén, Potosí and Buenos Aires. For this reason, soil and land use maps, previously developed separately for each municipal area, were combined in order to develop maps covering the whole river basin as a common area for environmental and development action.

In order to improve the understanding of the right fit of measures introduced by the PHES scheme, monitoring of land use changes and river discharge at different points of the river basin are continuously carried out to learn more about cause-impact relationships (see Figure 6.6).

Figure 6.6: Discharge measurement carried out for baseline assessment of hydrological service provision (Source: CASUR)

PHES scheme	Organizations	Respective tasks of organizations	Stakeholders involved
Gíl González (Hack, 2010)	Managing committee	Administration of the PHES fund, supervision and monitoring of land use changes / forest protection and ecosystem service provision	Service buyers (CASUR, municipalities of Belén), service providers, GIZ
	Technical committee	Capacity building for service suppliers, awareness raising and educational actions	Members of the Managing committee, CASUR, representatives of MARENA, MAGFOR, INTA, INAFOR and AMUR
La Golondrina (Baltodano, 2008)	Municipal council	Makes decisions on water management and provision as well as the administrative process of the payment mechanism	Mayor and three representatives of the municipal parliament
	Administrative council	Execution of the PHES and supervision of the water supplier, control of protected areas, promotion and amplification of the PHES including additional Service buyers and providers	President of the municipal water supplier and four members of the civil society (one school director and three service providers)
Estelí / El Regadío (Ardón Mejía and Barrantes, 2003; FAO, 2010c)	Water committee	Administration, management and maintenance of the water supply service, preservation and restoration of the river basin, execution of the PHES	NGOs (PASOLAC, MOPAFMA), local municipality, water users and service providers

Table 6.8: Examples of established organizations, respective tasks and stakeholders involved in selected PHES schemes (Author's elaboration

The river basin, as a geographical reference has further implications for the implementation of PHES concerning the institutional framework and stakeholder involvement. As the institutional interplay analysis suggests, implementing IWRM does not necessarily require the creation of new all-encompassing organizations, but rather a change of working practices to look at the bigger picture that surrounds each stakeholder's actions and to realize that these do not occur independently of the actions of others. IWRM implementation seeks to introduce an element of decentralized democracy towards the way water is managed with an emphasis on stakeholder participation and decision-making at the lowest appropriate level. In order to manage water resources sustainably and equitably all needs and uses of water, and therefore all respective stakeholders and relevant institutions behind those, have to be considered and integrated in the decision making process. This requires cooperation and coordination across policy sectors, stakeholder groups and often administrative units. This in turn, often results in problems of institutional interplay (see Section 3.2). Whereas traditional environmental policies in Nicaragua, basically municipal environmental management or development plans, have been elaborated within individual municipal boundaries without much consideration or consultation of different stakeholder groups, especially without the consideration of populations in areas far from the municipal capital, the implementation of PHES requires an assessment of the different stakeholders as well as a negotiation process among them to reach agreements. Water users as well as land users are considered for cooperation as either potential service buyers or service providers.

In the Gíl González PHES scheme, these stakeholder groups were, in a first step, identified and in a second step entered into a dialog with the aim of finding solutions for cooperation. Although the PHES concept has a conservationist background, it does not represent an explicit sector policy instrument, since it focuses on the cooperation or functional linkage of ecosystems, the facilitation of ecosystem services through certain land uses and beneficiaries from those services from all sectors. Hence, the logic of cooperation among service providers and buyers is not bound to predefined policy sectors. The PHES concept uses the strategy of "issue linkage", a strategy that makes the solution of an issue that is of concern to another actor dependent on the solution of an issue that is important to oneself (Mostert, 2003; Dombrowsky, 2007a), to bring up- and downstream parties together. In this case, the basis for this "linkage" is an intact ecosystem that provides the desired services and the solution to spatial fit as described above. This way the PHES concept facilitates a simultaneous discussion of issues formerly considered independent, e.g. environmental degradation, agricultural production and water supply, for a joint settlement.

The negotiating process in the Gíl González case as well as in other PHES schemes in Nicaragua was facilitated by the municipal authorities with the aid of donor organizations and ended in the establishment of organizations responsible for carrying out the PHES implementation and execution. Table 6.8 provides examples of organizations, their respective tasks and the stakeholders involved in selected PHES projects.

It is clear that PHES do not develop in the absence of prior environmental legislation, institutions and policies, but rather come up as an alternative or often a complement to traditional environmental policy instruments as the case of Nicaragua shows. Many attempts to allocate financial resources to environmental protection through different types of funds (e.g. National Environmental Fund, National Water Fund) were made in Nicaragua, but have failed in the past for different reasons. Problems of institutional interplay exist because institutions are poorly developed, e.g. the National Water Fund is still not supported by institutional or structural mechanisms for its implementation; the law on water use fees needs to be approved before fees can be charged. In addition, the National Environmental Fund, the National Cleaner Production Fund and the Fund of Private Forest Owners did not have the expected impact of channeling financial resources to the environmental sector (López Nolasco and Jiménez Otárola, 2008b). Contrary to "artificial" superimposed organizations that lack capacity and legitimacy (Lankford and Hepworth, 2010, cf.,), the organizations established through the PHES (e.g. management committee or the river basin committee in the Gíl González River Basin) are a result of the willingness to cooperate to address perceived problems. Institutional interplay was further promoted due to the participation of land users and ecosystem service buyers in the definition of potential areas of ecosystem service provision and mutual agreement on measures to improve ecosystem service provision that allow for compensation through payments.

Moreover, the actual political commitment at the local level, even where several municipalities are involved, is an important outcome of the PHES schemes established in Nicaragua. In all PHES schemes in Nicaragua, the local municipalities take an active role and contribute to the scheme with financial and human resources. Awareness building actions are a core element of PHES in order to involve service providers and buyers. Practical experience has shown that it was more difficult to involve providers than buyers in some schemes, while the opposite was true in other schemes. Poorer service providers, often because of lower opportunity costs, were more easily available as service providers than land users with higher incomes. Nevertheless, the view of upstream land users as exploiters of renewable resources is changing to one where they are seen as possible stewards for nature, using practices in synergy with ecosystem processes and functions (FAO, 2010a).

Experiences from the PHES scheme in the Gíl González river basins as well as other schemes in Nicaragua show that there has not been much success in involving the national public water supply company ENACAL as a service buyer, although it is present in all PHES schemes, except in the case of Río Blanco where the municipality took over the local ENACAL branch (Flores Barboza et al., 2011). In the case of Río Blanco, water users even expressed willingness to pay for the ecosystem service in addition to a water fee. In the case of Río Blanco, the municipally owned water supplier spent part of the PHES revenue to cover production costs (FAO, 2010d). In all cases, there is stakeholder involvement on both the supply and on the demand side although the benefits do not outweigh the costs of provision or real benefits are absent, e.g. upstream land users often participate even if they are not better off economically. This is sometimes because of a conservationist attitude or because of non-monetary benefits, e.g. inclusion in other development projects, compensations in kind.

Apart from solving problems of spatial fit and institutional interplay, the availability of information on natural resources in countries like Nicaragua poses an additional challenge for decision-making in the context of IWRM. Novo and Garrido (2010) point out that "[...] like many other data-poor countries, Nicaragua lacks a complete spatial and temporal water database". Therefore, a very important step at the beginning of a PHES implementation process is the assessment of the state of the ecosystem service and the main factors influencing its provision. In the case of hydrological ecosystem services a thorough assessment and mapping of soil types, land uses and topography is required. Quantification of baselines (e.g. river flow around the year, groundwater tables, water quality) of the current service status is also necessary to validate comparatively a service alternation through monitoring later on. This information provides the basis for identifying critical areas and conflicting land uses or other controversial actions (e.g. discharge of waste water) within the river basin. As already described above these actions facilitate the formalization of the river basin as the area of action (spatial fit), stimulate participation and cooperation with upstream land users (institutional interplay) and raise awareness for environmental problems in a productive way. The initial bio-physical assessment is often carried out by universities acting as consultants or by the National Institute for Territorial Studies. In the best case, this assessment is carried out in cooperation with the land users in a participatory way, as in the case of the Gíl González River Basin (FAO, 2010a).

Established organizations with PHES schemes generally consist of a managing organization that administrates the fund and that is responsible for establishing of legal contracts between service providers and buyers, and often also a technical organization that is responsible for monitoring of the service provision, e.g. by documenting changed land use and applied conservation actions, better also through direct measurements of services provided, and capacity building for the service providers (see Table 6.8). The established organizations facilitate coordination and cooperation in actions of conservation, e.g. the status of the conservation measures taken, the participation of service providers and water users as well as the service provision is communicated regularly. The status assessment, the establishment of a system of payments and the integration of new stakeholders can be realized by involving new stakeholders in the assessment actions and by implementing a system of payments to encourage the presence of potential additional service providers (FAO, 2010a). By

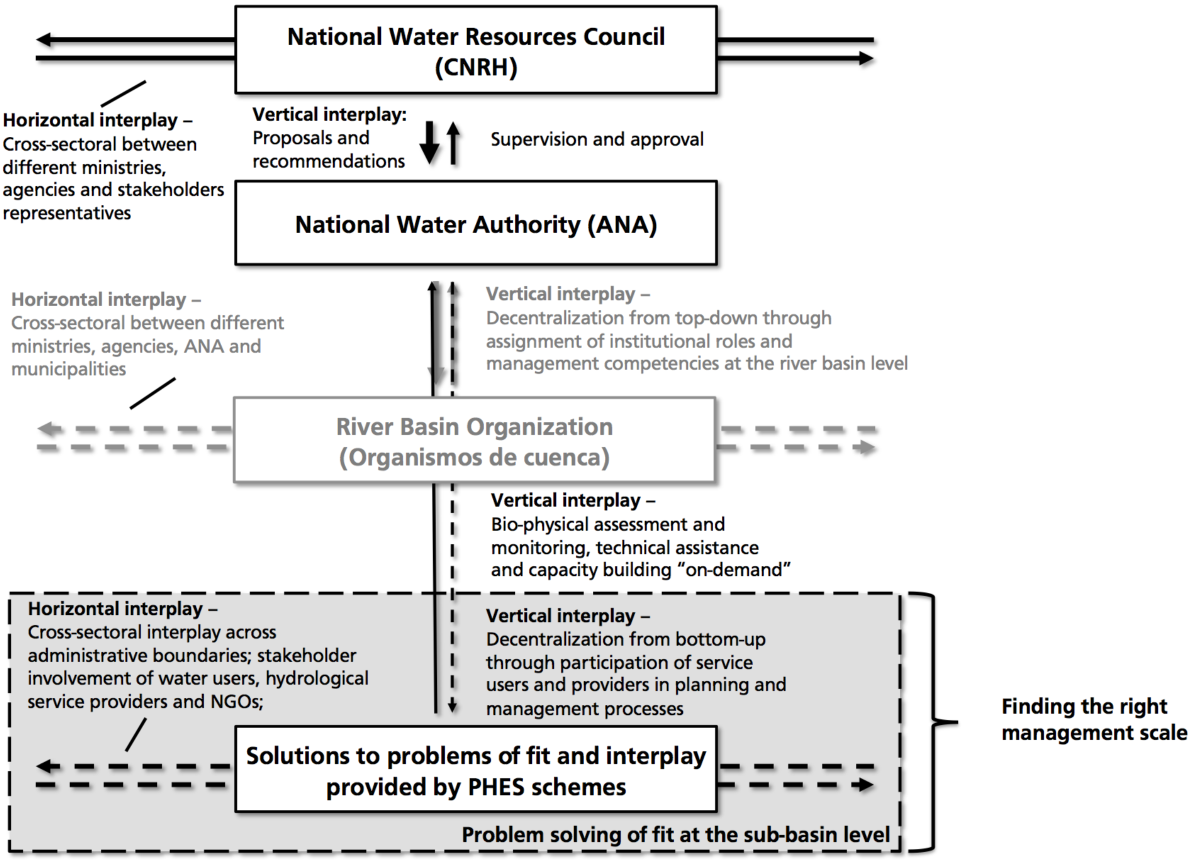

Figure 6.7: Functional role of PHES schemes in the Nicaraguan IWRM implementation process (Author's work)

doing so, this may serve as a promotional action in order to engage additional providers and buyers. This was done, for instance, in the case of the Gíl González PHES.

As a result of the PHES implementation process, in the next step a river basin plan is developed as a spatial planning action including a zoning of the river basin for different uses, e.g. in form of environmental management and development plans based on the river basin. The economic valuation of the considered hydrological ecosystem services is of particular importance within the PHES implementation. Starting at the demand side, the willingness of water users to pay is assessed based on the ecosystem service concept and their water use is quantified (at least roughly). The potential service providers can be identified based on the conflicting uses already assessed. Their cost to provide the service has to be estimated as well. A process of promotion and communication of the payment mechanism often accompanies this economic valuation (Flores Barboza et al., 2011). Capacity building in soil conservation measures and water saving actions for service providers and water users often are also included (FAO, 2010b). If the PHES is agreed upon, legal and administrative actions are required to implement the scheme, thus, municipal decrees are applied to establish a fund for the payments and one or more institutions to manage the PHES project. Although there is still no national law that regulates the establishment of PHES, several municipalities could put PHES in action by creating administrative institutions and rules for the payment mechanism (e.g. municipal environmental funds) through municipal decrees. This demonstrates that a feasible decentralization is possible. The municipal decrees and resolutions establish PHES funds, administrative and management institutions and provide room for public participation (Juárez Martínez, 2008).

In the context of the national IWRM process in Nicaragua, local PHES schemes, like the Gíl González scheme can possibly be a complement to achieve context-specific fit and interplay at the sub-basin level. As could be shown, through the involvement of hydrological service providers and beneficiaries in planning and management processes, horizontal interplay at the sub-basin level is achieved across sectors and administrative boundaries. The PHES management committee, in its facilitating role for spatial development planning through the establishment of protection and cultivation zones within the PHES schemes, has served as a pilot for the Inter-municipal River Basin Committee of the Gíl González River which was established in 2010 (CIRGG, 2010) and recognized by the ANA in 2012 (ANA, 2012). The principal objective of the Inter-municipal River Basin Committee is to coordinate the application of policies, plans, environmental actions and conservation financing mechanisms that contribute to the sustainable and shared management (CIRGG, 2010). The establishment of the RBC demonstrates that the organizational development of cross-sectoral and cross-administrative

IWRM structures from bottom-up can be promoted through a PHES scheme. The official recognition of the RBC by the ANA is one form of vertical interplay, another form of vertical interplay was achieved through the cooperation with other governmental agencies, e.g. INTA and INAFOR, in terms of technical assistance and support in monitoring land use changes. Furthermore, the PHES in the Gíl González river basin could overcome critical operational constraints of funding, capacity and informational gaps through co-management and co-financing by public and private stakeholders. Hence, in the case of the Gíl González sub-basin, current gaps in the institutional and organizational framework of IWRM in Nicaragua could be overcome. This is illustrated in Figure 6.7, the grey colored parts in the figure are formal IWRM elements which are still lacking.

Hence, PHES schemes can contribute to the national IWRM process by providing mechanisms for horizontal interplay at the management level and vertical interplay through the establishment of RBCs and *upward connectivity* to higher level sector organizations such as MARENA, MAGFOR, INTA and INAFOR. Moreover, the facilitation of horizontal interplay at the management level is directly related to the solution of the problem of institutional fit. Hence, PHES schemes can obviously provide for the appropriate scale, based on interrelated solutions of fit and interplay, of a management unit as it is perceived and expressed by locally involved stakeholders. The application concept of hydrological ecosystem services represents a means of achieving this.

In the context of the IWRM process in Nicaragua, PHES schemes bear the potential to complement the processes of horizontal interplay at the national level by contributing mechanisms for horizontal interplay at the management, i.e. operational level and vertical interplay from bottom-up. The vertical interplay from bottom-up may be readily picked up by a contrarian process of vertical interplay such as the creation of RBOs, as defined in the water law. Hence, PHES schemes can fill a significant implementation gap at the operational end of the IWRM process in Nicaragua.

6.4 Payments for Hydrological Ecosystem Service as part of a policy mix for IWRM

Considering the Gíl González PHES scheme as part of a broader mix of policy instruments, the instrument has proven to be more flexible and context-specific than other regulatory command and control instruments provided by the National Water Law. This context-specificity is particularly important to address problems of fit and interplay at the operational level. However, this is more due to the informative character of the PHES instrument, e.g. communicating the benefit flows within a river basin and how these are determined by different actors and actions, than due to strong economic incentives. Nevertheless, the instrument provides incentives, although not economically driven, to engage different upstream and downstream stakeholders in cooperation as well as in exchange of values and perceptions on hydrological functioning, i.e. service provision. Furthermore, the interaction of stakeholders in discussing the role of different land uses and their impact on hydrological ecosystem services facilitates the consideration of existing regulatory instruments. In the context of defining additionality and conditionality as well as in avoiding perverse incentives, a rule-making process takes place which concludes in agreements on which kind of land use is appropriate for specific sites, if it should be rewarded and how it should be rewarded. Thus, actions which violate existing regulations considerably, e.g. cutting down trees in riparian areas, may not be considered socially justifiable and it may be decided that restoration activities should not be rewarded beyond the provision of restoration materials. The rule-making process has further implications as it promotes the process of land use planning on the basis of river basins.

Although the PHES instrument basically represents an alternative, i.e. a complement to other forms of environmental regulation, the instrument can strengthen other existing regulation and promote further institutional development. In the Gíl Gonzalez PHES, for instance, the instrument promoted the introduction and diffusion of command and control policies. The question of what is illegal and should not be compensated has been discussed and led to existing formal rules being communicated and informal ones being contradicted. In several cases the identification of critical areas led to conservation actions that did not form part of the PHES scheme. For example, the protection of water holes and riparian areas in the Gíl González river basin in order to improve water availability during the dry season was not monetarily compensated but is still an agreed measure among scheme participants. Moreover, the implementation of the PHES schemes was accompanied by several awareness raising actions focusing on the upstream-downstream relationships and possible solutions for these problems (FAO, 2010b; Flores Barboza et al., 2011). Upstream land users could learn about the negative impacts they may have on downstream water users, but also got to know which actions are actually prohibited by law, e.g. use of riparian areas for agriculture or distortion of water source areas. These awareness raising and educational actions resulted in a variety of individual voluntary actions, e.g. the conservation and protection of riparian and headwater source areas with fences, water saving in irrigation systems and the installation of water meters in different households. Furthermore, through technical assistance from federal agencies the upstream land users learned about their on-site benefits from actions that result in improved service provision for downstream water uses, e.g. soil conservation measures that increase soil moisture and reduce erosion of fertile lands.

Table 6.9 summarizes formal rules which have been established in relation to the implementation of the Gíl González PHES. All of these legal initiatives have a cross-administrative character originating from the basin perspective introduced

Type of formalization	Description
Intermunicipal declaration	Declaration of the Intermunicipal Ecological Park "Laguna de Ñocarime" for the protection, conservation, restoration and sustainable management of the Ñocarime Lagoon
Intermunicipal decree	Decree on the protection of forests in Potosí and Belén
Intermunicipal decree	Decree on the restriction and prohibition on the use of agrochemicals in areas close to water bodies
Intermunicipal decree	Decree on the management of solid wastes
Intermunicipal regulation	Regulation of the organization and functioning of the Intermunicipal Committee of the Gíl González River Sub-basin
Municipal decree	Decree on the Institutionalization of Payments for Hydrological Ecosystem Services in the Municipality of Belén
ANA confirmation	Conformation of the Intermunicipal Committee of the Gíl González River Sub-basin

Table 6.9: Formal rules established within the process of PHES implementation and spatial planning in the Gíl González River Sub-basin (Author's elaboration

through the PHES scheme. Additional actors took part in this process. For instance, the NGO Foundation Ñocarime in promoting the establishment of the Intermunicipal Ecological Park "Laguna de Ñocarime" and the Nicaraguan Foundation for Sustainable Development (Funación Nicaragüense para el Desarrollo Sostenible, FUNDENIC).

Hence, the establishment of formal rules illustrates how municipalities together with representatives of the civil society are applying conservation laws, for instance, through the designation of protected areas (Municipal Park, Ñocarime lagoon). The declaration of the Ñocarime Lagoon and its surrounding wetlands as a protected area was the first protected area established by several municipalities jointly (Municipalities of Potosí and Buenos Aires, 2010). With the Decree on the Institutionalization of Payments for Hydrological Ecosystem Services in the Municipality of Belén, the municipality decided to formally introduce the PHES as a financial instrument for the conservation and the management of natural resources for all river basins forming part of its territory. Besides the Gíl González river basin, this includes the basins of the rivers Lajas, Tola, Escalante, Nahualapa, Ochomogo and Ñocarime (Municipality of Belén, 2011). The PHES instrument, the decree on the protection of forests and the declaration of protected areas represent the principal policy instruments. The establishment of the Intermunicipal Committee of the Gíl González River Sub-basin has formally introduced the development of Integrated Water Resources Management Plans (Plan de Gestión Integral de Recursos Hídricos) as a principal function of the committee. The committee consists of the mayors of the municipalities of Belén, Potosí and Buenos Aires, representatives of water users, service providers and governmental agencies.

How the PHES instrument has influenced different land and water uses is illustrated in Figure 6.8.

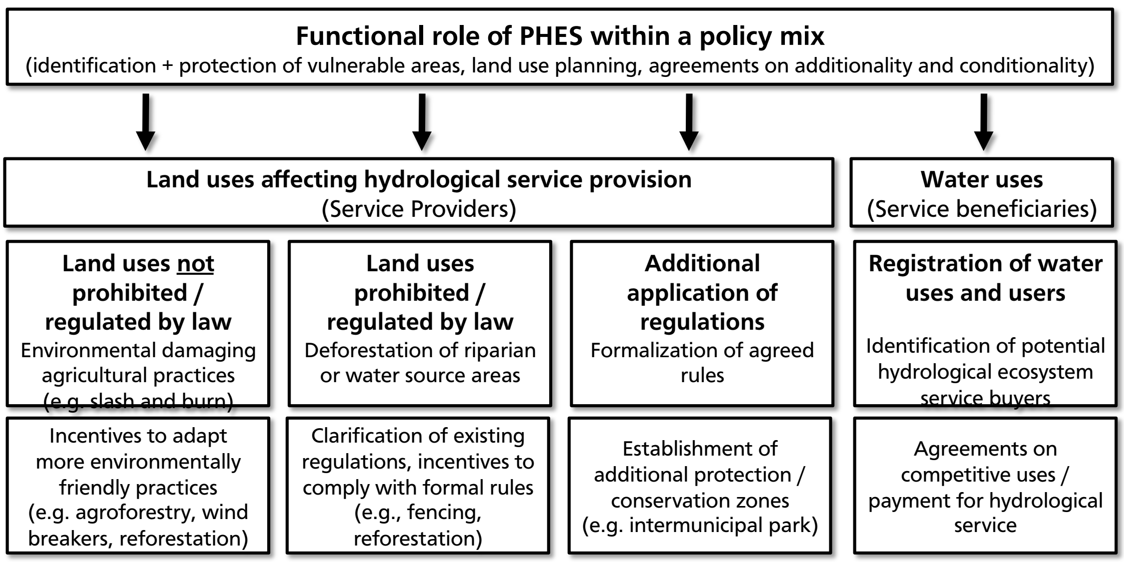

Figure 6.8: Illustration of the functional role of PHES within a policy mix (Author's work)

Implementation step/IWRM task	Outcome	Formal responsibility / means
Bio-physical cause-effect relationships between land use and service provision	Identification of provisioning zones, soils and land use conflicts, land use zoning on the basis of river basins	ANA or RBO through river basin management plans
	Water quality / availability assessment	ANA through water balances and quality baselines for all river basins
Socio-economic valuation of ecosystem services	Identification of land users (service providers) and water users (beneficiaries) WTP for service provision, WTA for changes in land use / service provision	ANA through the National Registry of Water Users
Negotiation between stakeholders towards agreements on land use	Agreements on land use and conservation measures to improve service provision through active stakeholder involvement	ANA or RBO through river basin management plans, RBC to achieve public participation
	Determination of conditionality and additionality	Different policy instruments regulating land and resource use
Continuous realization of agreed actions and payments	Continuous financing of conservation activities, payments for benefits derived from hydrological ecosystem services	Financing achieved through water fees

Table 6.10: Overlapping of PHES implementation steps with IWRM tasks in terms of outcomes. Formal responsibility is indicated in case the task would be carried out in line with national legislation assignments (Author's elaboration).

The functional role of the PHES instrument affects land and water uses in different ways. On the one hand, PHES schemes introduce incentives to voluntarily adapt more environmentally friendly practices where environmental regulation is absent and land uses affecting downstream water users negatively are not prohibited by any formal regulation. On the other hand, the instrument interacts with existing regulations for critical hydrological service provisioning areas, e.g. riparian areas and water source areas, through clarification of existing regulations not successfully enforced in the past. The instrument may provide incentives to comply with these formal rules (e.g., fencing, reforestation) once their importance for service production becomes apparent. Furthermore, as the Gíl González PHES has demonstrated, the scheme promoted the establishment of additional protection and conservation zones such as the Intermunicipal Ecological Park "Laguna de Ñocarime". Before the establishment of the scheme, local municipalities had not established any protection zones. Finally, another impact of the functional role of the PHES instrument is the registration of water uses and users through the identification of potential hydrological ecosystem service buyers so that the scheme can function. In developing countries like Nicaragua, this information is often lacking and represents a significant obstacle to the development of water fees and control of resource use. Establishing the PHES scheme in the Gíl González river basin meant that all commercial water uses and users could be identified. Moreover, competitive water users are sharing a common goal in improving the provision of a service they all depend on. In the Gíl González PHES, this has led to agreements among competitive water users on how to use water more efficiently (AMUR, 2013; Sánchez et al., 2013).

According to the National Water Law, the ANA is responsible for creating RBOs and RBCs. Moreover, either the ANA or, if established, a corresponding RBO is in charge of carrying out bio-physical and socio-economic assessments in order to formulate Integrated River Basin Management Plans. However, at the moment the ANA does not have the capacity and resources to realize these important tasks. Instead, the ANA's role in the IWRM process at the operational level is limited to a legally advisory one through the confirmation of RBCs which were established independently from the ANA's activities. This is documented in the case of the Gíl González PHES scheme and also in other projects, for instance, in the elaboration of a River Basin Management Plan for the Coco River in the north of Nicaragua (Baca et al., 2012). For the moment, the ANA has to focus on the IWRM process at the normative and strategic level, acting as a water ministry in progressing and moderating the national IWRM dialog at the national level. More important than the *arbitrary* creation of RBOs from top-down without specific regard to local context is the establishment of laws and further institutions, e.g. the National Registry of Water Uses or the establishment of the National Water Fund, which are needed to take the IWRM process to an operational level.

The implementation and execution outcomes of PHES schemes in Nicaragua assume several actions and establish means which are formally under the responsibility of the ANA, a RBO or a RBC. Table 6.10 summarizes how these outcomes relate to formal responsibilities according to the water law. Because of the limited capacity of the ANA to assume responsibilities itself and in the absence of RBOs and RBCs, these outcomes represent an important input to the operationalization of IWRM in a bottom-up manner. The confirmation of the RBC of the Gil Gonzalez river basin, which is a result of the PHES scheme development, through the ANA demonstrates that the ANA is actually open to this bottom-up initiative.

6.5 Summary

The Nicaraguan IWRM process is typical for developing countries in many aspects. The process started during the 1990s when the issue was quite recent on the international political agenda. The development of the National Water Action Plan in 1998 was strongly supported by the international donor community and marks the beginning of the IWRM process in Nicaragua. However, it took almost another decade until the National Water Law was finally passed and the first steps towards an enabling environment as well as the (re)assignment of institutional roles was implemented. Despite progress at the national level in terms of strategic and normative enactment of IWRM, the operationalization of IWRM at the management level has not been achieved. The establishment of the National Council of Water Resources and the National Water Authority (ANA) represents an important step in improving horizontal interplay at the national level and prioritization of integrated management approaches for water resources, however, the multitude of tasks still to be done at the strategic level of IWRM, e.g. the establishment of the National Registry of Water Users or the passing of complementary laws to achieve appropriate financing of the operational IWRM process, as well as limited resources and capacity, disable the operationalization of IWRM at the river basin level.

In these circumstances, the PHES instrument can make remarkable contributions to IWRM operationalization at the local level. The National Water Law actually introduces PHES as a management instrument in order to promote the compensation of conservation measures and the rational use of water resources. The water law assigns the task of PHES implementation to the ANA, in practice, however, PHES schemes are implemented and executed without contribution from the ANA. These locally organized schemes, often initiated by international donor organizations and local NGOs, contribute in many different ways to the operationalization of IWRM through the adoption of tasks formally corresponding to the ANA, RBOs or RBCs as discussed in the previous section.

The main goal of water resources management is to make the best use of the water resources available. Externalities and trade-offs are inherently part of water resources management making the coordination of different water uses and of those actors that have an impact on water resources a due requirement. Since water resources management is basically an allocation task for a finite natural resource, which is provided as a natural input (precipitation) and initially influenced by the characteristics of the land on which it falls, considerations of different degrees of hydrological ecosystem services provision from different types of land uses appear very much straightforward. The IWRM concept emphasizes the integration of different policy sectors within the natural ecosystems of river basins as a spatial fit solution. The coordination and cooperative challenges of institutional interplay which this implies have been addressed repeatedly by IWRM promoters. Nevertheless, attempts to overcome these obstacles to IWRM implementation through new policy instruments that explicitly incorporate approaches to solve them are still an exception. Water managers and agencies often do not consider the value of hydrological ecosystem services resulting in the prevention of ecosystem degradation. Water management has traditionally focused on single factors directed more toward individual concerns such as water pollution control, water supply and allocation, and specific targeted water-use sectors, rather than considering them collectively. An ecosystem services approach towards water management instead focuses on the broader goal of balancing and sustaining ecosystem services as a prerequisite for meeting these (and other) sectoral needs.

In this context, the experience of PHES implementation in Nicaragua could demonstrate feasible solutions to the problem of spatial fit at the sub-basin level. The problem of institutional interplay is solved by a plain problem-orientation and solutions offered by the ecosystem service concept. This has facilitated cooperation across different sectors and the creation of suitable institutions in a “fit for purpose“ way. The PHES implementation led to important improvements in the knowledge base on natural resources, considerations of suitability of land uses and existing laws on environmental and natural resource use. The availability of such a knowledge base is a prerequisite for sound decision-making in water resources management.

If an integrated river basin planning and management becomes legally institutionalized, e.g. through a management unit within a PHES project carrying out the tasks of a river basin committee or through linking PHES to statutory land use planning, a sustained development of the river basin can be enabled. The Gíl González PHES demonstrated that the instrument can promote site-specific and problem-related institutional arrangements keeping transaction costs as low as possible and eliminate the establishment of new or additional institutions. Recognizing the success of the Gíl González PHES, the neighboring municipality of Belén, the municipality of Tola, started to replicate the PHES scheme of the Gíl González in another river basin with touristic installations as service buyers (AMUR, 2013).

However, it remains to be seen how basin committees created through the National Water Law, i.e. the ANA, will affect these institutions created within the PHES process. The implementation and execution of PHES schemes is not a substitute for IWRM, but because of the tasks that need to be done within their implementation and later on in the continuous application, PHES can potentially contribute more to the IWRM process than other economic instruments. While PHES apply to hydrological ecosystem services within a river basin or at least part of it, the IWRM process includes further geopolitical levels. In this context PHES may improve the bottom-up process of IWRM implementation only half way, IWRM needs to complement this process from top-down with national policies and general political agenda setting for IWRM. In Nicaragua, for instance, this is intended with the National Water Law and the National Water Authority. Connecting local PHES projects to the national top-down IWRM process remains a challenge for the future. PHES, however, also have limitations that should not be neglected and many of their positive “side-effects” (promotion of the river basin as area of action, stimulation of decentralized and participatory management of water resources, communication of river basin processes) are possibly only due to corresponding project design and small-scale projects. If the natural environment is recognized not only as a provider of ecosystem services, but also simultaneously as threatened by pollution and over consumption, the basic needs of intact ecosystems may be integrated by PHES concepts as well. In particular, the consideration of over exploitation and downstream pollution control has remained largely unaddressed in the projects reviewed. In PHES schemes where payment has been implemented on a per volume basis of the water consumed, there is at least a small incentive to save water but not explicitly for the environment.

7 Conclusions

This dissertation has documented, analyzed and interpreted the state of knowledge of the global IWRM implementation process. Based on the assessment of global implementation reports and the scientific discourse of two decades general problems of implementation and specific constraints to the process in developing countries could be identified. Moreover, present IWRM trajectories and recommendations towards improvements of implementation and operationalization could be derived from scientific and practitioner's experience. Hence, the principal contribution of this dissertation in this basic problem analysis is a clarification of the problem statement of IWRM implementation in developing countries and the provision of guiding principles for improvements.

The solutions approaches identified were further conceptualized methodologically by applying the theoretical concept of institutional fit and interplay. In the following, this dissertation asserted that problems of institutional fit and interplay need to be solved interdependently with due regard to specific operational constraints of local context in developing countries in order to successfully implement IWRM at an operational management level, i.e. operationalization of IWRM. Hence, this dissertation compared different policy instruments to achieve this based on an actor-centered incentive approach. It was argued that improvements require a mix of policies with specific instruments to enhance institutional fit and interplay suitable for context-specific operational constraints. This dissertation could expose that traditional command and control approaches for IWRM operationalization are insufficient and complementary instruments are necessary to address the prevailing implementation gaps. The instrument of PHES was identified as a potentially suitable policy instrument because it combines several beneficial characteristics of communication / diffusion and economic instruments with characteristics of collaborative agreements. Applying the concept of institutional fit and interplay in the context of operational constraints of developing countries to identify suitable policy instruments for improved IWRM operationalization, as done in this dissertation, is a particularly novel approach.

In order to assess the potential contributions of PHES schemes to the operationalization of IWRM, the concept of ecosystem services and their valuation as its theoretical basis, was examined concerning the solutions it provides to address the governance challenges of institutional fit and interplay. In this context, the concept of hydrological ecosystems services their valuation process was addressed specifically. This dissertation could contribute to understanding how the application and valuation process of hydrological ecosystem services inherently define spatial relationships between potential service providers and beneficiaries based on functional ecosystem linkages. Hence, it was illustrated how the concept can positively influence the identification of appropriate context-specific scales for operational IWRM as a result of fit and interplay interdependence. Additionally, a cross-sectoral and cross-jurisdictional integration effect of the concept could be acknowledged. The concept of hydrological ecosystem services has not yet been considered in relation to problems of fit and interplay in IWRM implementation as presented in this work. Hence, this dissertation could provide additional insights in this regard.

In a further theoretical analysis, this dissertation documented, analyzed and interpreted the state of knowledge of the economic conceptualization of the PHES instrument. This dissertation could bring forward supporting arguments for a less market-based and therefore a stronger multi-faceted incentive-based interpretation of PHES. Based on a broad meta-analysis of global and regional PES scheme assessments, the principal characteristics of the instrument and its implementation could be identified. A comprehensive instrument characterization of this kind is a further significant contribution of this dissertation which has not been done yet to a similar extent. The characterization and instrument assessment provides important insights with regard to the typical roles of different actors and the provision of incentives for behavioral change towards IWRM. Moreover, it contributes to understanding how the instrument can potentially address institutional challenges of IWRM operationalization in the context of general operational constraints. However, locally user-(co-)financed PHES schemes as a particular type were identified as especially conducive to engaging stakeholders and to promoting public participation.

Finally, the role of the PHES instrument in the context of a national IWRM process was assessed based on an empirical example taken from Nicaragua. This dissertation provides a comprehensive documentation of the national IWRM process in Nicaragua and its principal implementation gaps. The generalization of implementation gaps and specifically operational constraints made before could be confirmed for the Nicaraguan IWRM process. Moreover, the shortcomings of a formal top-down implementation approach based on command and control instruments alone could be highlighted as well. Further valuable findings can be derived from this dissertation for other developing countries with a similar IWRM process through the analysis of contributions of locally user-(co-)financed PHES schemes to solve the problems of institutional fit and interplay in Nicaragua. Additionally, this dissertation revealed how the PHES instrument fits into an existing policy mix in Nicaragua and how it interacts with other regulations. Hence, this work provides guidance on how to improve context-specific fit and horizontal interplay at the operational level of IWRM as well as on how to complement the primarily top-down directed IWRM implementation from bottom-up. Hence, this dissertation could document that the

PHES instrument is more than a tool to finance nature conservation. Indeed, it could be shown that the implementation and execution process of PHES schemes fulfills several other tasks which are essential for the operationalization of IWRM.

In the following, the general aims of this dissertation, regarding the objective of investigating how and to which extent PHES fits in the IWRM process and what role PHES can potentially play to further operationalize IWRM, are addressed specifically by answering the questions defined in the introductory chapter.

Why is IWRM still not widely implemented, despite having been promoted strongly by the international community over the past three decades?

The general problem analysis of the global IWRM implementation process in this dissertation could reveal that while IWRM is increasingly implemented at the national level through new legislation and organizational development corresponding to IWRM objectives, the operationalization of IWRM at the river basin level lags far behind. Blueprint, often all compassing, approaches of IWRM implementations were identified as a major problem. These 'standard packages' do not consider special contexts and problem situations in specific river basins and therefore fail to find the right scale for management and to engage the relevant stakeholders. Besides the complexity of the top-down comprehensive integration task, additional constraints in form of scarce financial resources, capacity and weak governmental performance pose further challenges to IWRM operationalization in developing countries. Hence, many scientists and practitioners advise to focus on more pragmatic, expedient or light approaches toward IWRM implementation, especially at the operational level, instead of overly comprehensive and complex approaches. Being more pragmatic does not imply discarding integration, instead integration should be based on specific problem contexts resulting from natural and human system interactions. Thus, applying IWRM principles to a specific problem context as an interpreted form of IWRM is assumed to be a more promising approach to start IWRM operationalization.

A focus on interactions of human actions through linkages provided by the natural system presents a useful means to achieve context-specific integration. Moreover, if benefits can be derived from these interactions and linkages, IWRM operationalization is facilitated. Context-specific approaches are supposed to provide more flexibility to address operational constraints and political problems. As a result of these insights, an adaptive IWRM concept, based on poly-centrism and social-ecological-systems theory, was developed. Adaptive IWRM emphasizes solutions based on the linkages of social (human) and ecological (natural) systems. A general principle of adaptive IWRM is to base management actions on a process of continuous learning about natural and human system interaction. Being aware of the need for a more adaptive approach, local action contexts are seen as fundamental when effective changes in water resources management are to be achieved. To put IWRM into context, this means moving from global, exogenous 'solutions' to local, endogenous plans of action. Thus, the key to improving IWRM implementation is to address the institutional and political challenges typically encountered when implementing integrated approaches. These challenges are problems of institutional fit and interplay, lack of participation, equity and accountability, as well as the general mismatch with needs and conditions in specific places. Therefore, a problem and people-orientated approach is recommended that avoids the pitfalls of the 'one-size-fits-all' attempts which often characterize mainstream implementation.

In order to improve IWRM implementation from bottom-up, the importance of social-ecological-system's interactions is highlighted. Both scientific communities (the one focusing stronger on the human system and the one stressing the natural system) acknowledge that a context-specific and problem-oriented approach is necessary to achieve this. Since humans are the central drivers of ecosystem change but at the same time firmly dependent on the goods and services that ecosystems provide, this interdependency has to be reflected in IWRM as well.

This dissertation has contributed to an improved understanding of the global IWRM implementation process and it has highlighted the status and principle problems of IWRM implementation in developing countries. Moreover, this dissertation has elaborated the main arguments and findings of the contemporary scientific discourse on the concept. Finally, this work could identify the current development trajectories of the concept and derived prevailing implementation recommendations to improve IWRM operationalization. These contributions to the problem analysis of IWRM implementation in practice ended in a generalized problem statement which constitutes the basis for the following methodological conceptualization of IWRM implementation.

What are the principal implementation gaps and core problems of the concept's operationalization in developing countries?

The methodological conceptualization of the IWRM implementation problem applied in this dissertation could demonstrate that, in order to operationalize IWRM at the river basin level, the principal problems that need to be resolved relate to achieving context-specific institutional fit and interplay. Institutional fit corresponds to the specific properties of the ecosystem, i.e. the natural system, to be managed and interplay corresponds to the interaction of institutions and users of the resource, i.e. the human system. Both, fit and interplay, interdependently define the right management scale, thus, integrating the specific natural and human system context. The common implementation gaps of IWRM incorporate the issues of institutional fit and interplay as well as additional issues of prevailing operational constraints. These operational constraints include funding, capacity and informational gaps which are of special importance in the context of IWRM operationalization in developing countries. Specific policy instruments are needed to improve IWRM operationalization in developing countries. On the one hand, there are several environmental policy instruments, often representing command and control instruments. On the other hand, these instruments, despite their advantages, have important limitations with regard to common operational constraints. Moreover, conventional environmental policy instruments do not specifically address the problems of fit and interplay and their interdependence. Thus, solutions to operationalize IWRM have not yet had the expected success.

The typical IWRM implementation which addresses problems of institutional fit and interplay separately, basically through political and / or membership linkages, was not able to operationalize IWRM in developing countries. This dissertation contributed to document this failure and provided suggestions to finding ways how to solve problems of fit and interplay interdependently while taking operational constraints into account in order to find the right scale for operational management based on natural and human systems' characteristics.

What are the specific requirements for instruments to improve IWRM implementation?

Different policy instruments have different advantages and disadvantages. Traditionally, environmental regulation has relied on command and control instruments. However, economic instruments have gained importance in the past because of their flexibility in achieving environmental goals, even beyond standards.

Policy instruments differ in terms of their rationale and implementation mechanisms but also with regard to the assignment of roles to different actors, e.g. more active or more passive roles, and the provision of incentives to encourage actors to change their behavior. In practice, however, combinations of different policy instruments aiming at combing individual instruments' advantages and overcoming disadvantages proved to be most effective, especially in the context of IWRM where multiple actors from different sectors are addressed and stakeholder involvement as well as public participation is important for success. In developing countries, for instance, experiences with the application of command and control instruments for environmental regulation alone have had very limited success. This policy failed for a number of reasons including limited resources, restricting a proper enforcement of the instruments for environmental regulation, problems of corruption and conflicts with informal rules. Moreover, in societies where environmentally damaging land use results from subsistence farming practices, it is hard to enforce rules with sanctioning consequences.

Command and control instruments have the advantage of being clear and easy to understand, but lack the flexibility and ability to provide incentives for stakeholder involvement and participation. However, command and control as well as classical economic instruments influence actors' behavior through changes in the external structure of objective options and constraints but they have limited influence on actors' internal structure responsible for behavioral motivation to act in a certain way. The internal structure of actors' decision-making is often influenced through communication and diffusion instruments. Collaborative agreements represent policy instruments that are able to influence both, the external and the internal structure. Relying on command and control instruments alone will not lead to improvements in institutional fit and interplay at the operational level.

Based on the methodological conceptualization of IWRM implementation gaps as problems of institutional fit and interplay in a context of operational constraints, this dissertation contributed to assessing how different policy instruments can improve IWRM implementation as part of a policy mix. The importance of taking a more actor-centered approach in providing incentives for actors to engage in institutional design and decision-making was highlighted. Hence, the requirements of complementary management instruments to improve IWRM operationalization were summarized as follows: (a) being flexible enough to match human and natural system context, (b) addressing ecosystem characteristics and interactions of humans with the natural system at the right scale, (c) providing incentives for cooperation and collaboration across sectors and administrative boundaries at the identified scale, (d) encouraging social learning, self-regulation and participation, and (e) embedding management instruments in a diverse institutional and social system (compatibility with existing regulations).

A top-down approach from the national level may provide a framework for greater collaboration among principle stakeholders through formal guidelines, but cannot ensure collaborative outcomes *per se*. Critical factors for active stakeholder engagement are building trust, working relationships and incentives to negotiate in order to evolve a common interpretation of the problem on which collective action should be focused. To achieve this, more flexibility is necessary to address different contexts and different problem perceptions, e.g. of costs and benefits from certain management actions.

PHES have recently become very popular in developing countries in order to overcome the disadvantages of regulatory command and control instruments. These instruments are based on the ecosystem service approach and allow environmental problems to be addressed more flexibly and context-specifically. As part of a broader policy mix among existing policy instruments, they seem to be a promising complement to traditional instruments as they combine elements of economic and communication instruments with collaborative agreements.

Do the ecosystem services concept and PHES, in general, bear potential to contribute to improvements in IWRM implementation?

The ecosystem service concept is increasingly applied as a governance framework to communicate the socio-economic dependence of humans on natural ecosystems. The concept highlights the relationship between ecosystems as service providers and humans as beneficiaries. This implies a spatial and temporal cause-effect dependence, thereby responding to the interrelated problems of fit and interplay. Moreover, human activity determines the provision of hydrological ecosystem services through a specific land use as well as the demand for services through water uses. Hence, the ecosystem service concept integrates land and water uses across sectors and administrative boundaries in a context specific way. The valuation of ecosystem services allows identifying social preferences for the provision of these services and may look at the supply and demand side. Moreover, the valuation process can be critical for identifying priorities in decision-making on land and water uses and communicates different value perceptions. Additionally, previously unknown or neglected use and non-use values of ecosystem services can be revealed. In the case of hydrological ecosystem services and their valuation, the spatial relationship of provision and demand becomes apparent and integration may occur accordingly.

The concept of hydrological ecosystem services is especially appealing because it provides a variety of spatial, temporal and scale-dependent relationships. Besides addressing these relationships in an ecological sense, the concept also addresses them with regard to socio-economic and cultural aspects in the context of analysis and valuation as well as supply and demand. So far, spatial and scale aspects of human and natural systems' interactions have merely been considered in the ecological context alone. The hydrological ecosystem service concept expands this perception by linking ecological, spatial and scale dependent characteristics through institutional mechanisms across spatial scales of service provision and service benefits in order to include, i.e. integrate, and activate different stakeholders of land and water uses to promote (better) collaboration between them. The understanding of ecosystem services as a result of ecosystem functions stemming from ecosystem structures and processes provides for new perspectives and consideration of scales. Ecosystem structures and processes function at different spatial scales ranging from very local over regional to global scales where they manifest themselves in different manners. Consequently, the assessment and valuation of hydrological ecosystem services requires a spatially explicit methodology. Thus, the application of the hydrological ecosystem services concept implies considerations regarding the scale of provision and the impact of benefits as well as appropriate reference units for provision and demand. Moreover, the distribution of roles of service providers and beneficiaries defines spatial scales of actions based on underlying bio-physical cause and effect relationships. Different provider-beneficiary constellations then result, dependent on the type of (hydrological) ecosystem service considered. This provides additional flexibility for the identification of appropriate management scales. This dissertation could illustrate how the concept of hydrological ecosystem services can contribute to the critical process of identifying the right management scale based on the functional scale at which ecosystem processes operate and the decision-making of key stakeholders should take place.

In the context of ecosystem-based land use planning, the ecosystem service concept has special significance because it directly addresses the relationship between the natural and the human system by connecting the social definition of space (area of perception and action) with the physical definition of space (distances, relative positions, boundaries). Hence, the hydrological ecosystem service concept provides a suitable framework to conceptualize the link between the natural system and how different actors value and use it. The variety and values of ecosystem services, thus, can be assessed at a scale relevant for the ecosystem service provision and political action to influence them, e.g. a river basin or a specific part of it where functional linkages and benefits show considerable interaction.

Besides mapping service provider and beneficiary relationships based on ecosystem functioning, the hydrological ecosystem service concept can identify how different actions affect the provision of ecosystem services and how those actions affect different beneficiaries. Hence, trade-offs can be considered in a broader context of potential on-site and off-site costs of provision as well as benefits from delivery. Providers and beneficiaries, as key stakeholders identify and articulate the ecosystem services which appear valuable. This valuation can go beyond monetary use values, but often money is used as a common currency for comparison in order to aid decision-making. In the context of valuation, it has to be clear that value expressions can be highly volatile and are generally subjective. Thus, a specific expression of value has to be regarded as being strongly dependent on the body that articulates this value. Therefore, ecosystem services are valued according to specific benefits instead of the ecosystems themselves or their structures, process or functioning. One way to take action against the threat of the instrumentalization of nature for specifically valued services at the cost of others can be to improve the underlying processes and structures of ecosystems through increasing biodiversity and system resilience as the basis for ecosystem functioning.

Despite its pitfalls, the valuation of ecosystem services allows the development of payments for ecosystem services mechanisms as a useful complement to conventional command and control instrument to operationalize IWRM. Especially in developing countries and in contexts where command and control instruments are not feasible, PHES reveal significant advantages in providing a coherent context to incorporate stakeholders and complex biophysical processes into a consistent, learning-based management scheme. The PHES logic provides incentives to address specific functional provider-beneficiary relationships, thus, crossing sectoral and administrative boundaries. On the one hand, PHES schemes offer a framework to define appropriate scales for management and, on the other hand they provide incentives for beneficial behavioral change of key actors. Moreover, the implementation process of PHES is actor-centered in order to solve functional cause-effect linkages and allows participants to enter into a rule-making process.

In a context where ecosystems are increasingly dominated by human uses and their alternation, a focus on ecosystem functioning seems particularly useful to define a desired ecosystem state. Often a reversal to a pristine state, i.e. a state before human impact, is illusive and trying to achieve improvements in ecosystem structure and processes that facilitate important ecosystem functioning (e.g. self-regulation of water quality) is more meaningful. The role of biodiversity to achieve this is often extraordinary. Thus, defining desired ecosystem conditions (reference) based on a resilience and flexibility considerations equilibrium instead of a static stability could shift attention from the restoration of the pristine, to the achievable and functional aim of rehabilitating stressed ecosystems. In the EU this way of thinking has led to a process of revision of the 'static' goal of a good ecological status towards a status definition of the ecosystem's potential for ecosystem processes or structures that provide the relevant ecosystem services (cf., Josefsson and Baaner, 2011). Considering potential ecosystem structures and processes as the basis for ecosystem services of a desired ecological status possibly yields greater benefits than a focus on biological quality elements in isolation from the broader natural system and its role within this system.

Payments for Hydrological Ecosystem Services (PHES) have gained significant popularity in the field of natural resources management in developing countries. The wide-spread and increasing application of PHES in Latin America, Asia and Africa as well as in North America and Australia documents the high expectations in the instrument. Besides a number of national schemes, especially locally organized PHES schemes are growing in number.

With regard to IRBM, this dissertation has demonstrated the benefits that a typical implementation process of a PHES scheme can bring about. These benefits include the identification and engagement with stakeholders at management level relevant for IWRM operationalization, the associated characterization and assessment / documentation of hydrological ecosystem services, the establishment of agreed cause-effect relationships and definition of objectives, the agreement on and implementation of selected measures to achieve objectives (e.g. service provision and on-site benefits), and also the monitoring and continuous evaluation of achieved land use changes and service provision. A fundamental element of PHES schemes is the engagement with relevant stakeholders representing providers, protectors and users of the ecosystem services. It is their interaction and interchange of value perceptions that determines the principal scheme design and, thus, the rule making process for actions to be promoted. In conventional IWRM operationalization, there is no agreed method for selecting stakeholders, e.g. to form part of an RBO or RBC, instead stakeholders are assigned by prescription in a formal top-down manner. In contrast, in a PHES scheme, stakeholders are engaged based on site-specific functional linkages of natural and human system interactions. There is an inherent motivation of PHES schemes to involve all relevant stakeholders in the participation. This implies actual actor-centered incentives for participation. Moreover, agreements on actions to improve service provision in the form of shared decision-making increases local ownership among actors. Hence, local actors can contribute to decision-making and gain political importance / power with their knowledge, e.g. on the land uses related to service provision.

At the beginning the promise of PHES schemes has been on efficiency gains (through better targeting of conservation measures) however, the experiences with PHES schemes documented in this dissertation could highlight several other benefits of at least equal importance. An important benefit is the possibility to align local aspirations (bottom-up) with top-down regulatory priorities as well as the engagement of stakeholders and empowering them to act to improve their communities, often through trusted intermediaries.

Furthermore, (because of their focus on functional linkages) PES schemes bear potential for the integration of cross-sectoral issues beyond land and water uses. The ecosystem service concept and payment schemes based on it can be perceived as a linking concept between water management and other sectors such as nature protection or climate change mitigation and adaptation for instance. Hence, payment schemes can be a focal point when linking different national and local environmental policies for flood protection, marine ecosystems, reduction of diffuse pollution and biodiversity strategies, just to name a few. PES schemes for hydrological ecosystem services have successfully been linked with objectives like improving the beauty of a landscape for tourism, sustainable farming, carbon sequestration and biodiversity conservation.

Despite promising advantages, PHES schemes are also confronted with significant challenges. One of the most important challenges is related to the uncertainties with regard to the identification and measurement of ecosystem services. Often the necessary data to derive past, present and future cause-effect relationships is missing. Apart from unclear cause-effect relationships, spatial and temporal scales of service provision are also often unknown. Consequently, actions to improve service provisions are basically assumed and based on shared beliefs. At best, this is clearly communicated among stakeholders in order to avoid disappointments. In order to enter into a learning process, it is important to monitor compliance with agreed actions as well as monitoring and measuring changes in service provision. This monitoring and measuring is also important to maintain the interest and good will of stakeholders. The effective engagement with stakeholders, the monitoring and measuring of actions and outcomes requires resources which will reduce the available funding for payments. Hence, the spending on transaction costs for stakeholder engagement and monitoring has to be balanced with the spending on payments. Here, the stakeholders involved have to set their priorities very carefully. For local PHES schemes, financial support from the national policy level can be very helpful. Complementary national payments can be justified when national priorities or political interest can be safeguarded within a scheme. Avoided deforestation, for instance, can be of national interest for climate change mitigation, besides being locally important for the provision of hydrological ecosystem services. The ability of PES schemes to link different sectoral policies, thus, represents an advantage with regard to the integration of top-down outputs and bottom up desires.

In developed countries, PHES are already a cost-effective alternative to reduce water treatment cost where diffuse pollution problems are present (e.g. in the USA) and are also increasingly recognized as a way to co-finance IRBM and a means to involve relevant stakeholders (e.g. in the UK, Australia and Canada). There is potential to link different environmental policies (e.g. biodiversity and flood protection, sustainable agriculture and tourism). There is also cost-effective achievement of good ecological status, identification of environmental and resource costs for the cost recovery for water services in the context of the EU-WFD.

In Art. 9 of the EU-WFD, water users are asked to contribute to the cost recovery of water services including resource costs[1] and environmental costs[2]. Furthermore, in Art. 11 the WFD calls for cost-effective measures to achieve a good ecological status. The inclusion of resource and environmental costs for water services aims at internalizing external effects of economic activities by charging water service uses the *full* cost. However, environmental and resource costs were not sufficiently defined in the WFD. Moreover, there are no methodological guidelines regarding their practical assessment and inclusion in the economic analyses of the WFD. Hence, the hydrological ecosystem services concept and PHES are now increasingly considered for guidance and as a practical means for implementation. The hydrological ecosystem services concept can also potentially contribute the definition of water services in a broader context of benefits derived from water and related land ecosystems. At the moment, the consideration of water services for which costs including environmental and resource costs should be recovered are limited to drinking water provision and waste water treatment for which pricing systems already exist. However, water services can be interpreted in a much more comprehensive sense as general hydrological ecosystem services, thus, benefits that society derives from ecosystems. This could lead to the inclusion of water services which have been traditionally 'free of charge' in many countries, e.g. water for irrigated agriculture, industry and even navigation and hydro power. Thereby, the financing of the sustainable provision of these services could, on the one hand, be shared between a larger number of users and, on the other hand, additional actors can be integrated in the policy process. The ecosystem services approach and PHES are now applied experimentally, for instance, in the context of the EU-WFD implementation in the UK, promoted by the Department for Environment, Food and Rural Affairs (DEFRA) of the British government.

In developing countries, the contribution of PHES schemes to finance IRBM is even more important than in developed countries as resources are especially scare. The improved engagement of stakeholders at an operational level of IWRM through PHES schemes is equally beneficial in developing countries. Beyond this, PHES schemes can reduce capacity and informational gaps through the identification of bio-physical cause-effect relationships between land use and service provision (baseline assessment) and the monitoring and measuring of outcomes. The bio-physical assessment within scheme establishment is an important contribution to the land use planning process based on river basins and cadastral development. Moreover, PHES schemes interact with existing formal rules and can improve their application.

The interaction with existing formal and informal rules is a result of the negotiating process between service providers and beneficiaries. In this process agreements have to be reached on how the provision of hydrological ecosystem services can be maintained or improved. In the negotiating process, it is important to agree on terms of conditionality and additionality of service provision. This often implies a discussion on which actions should be implemented and how they should be rewarded. In this rule-making process compliance with existing regulations, e.g. to protect riparian areas or steep slopes, becomes an issue and stakeholders agree on uncompensated compliance. Through targeted incentives, PHES can provide short-term tangible benefits to land users in order to promote long-term behavioral changes towards IWRM. Moreover, compared to other policy instruments, PHES can bring about a transition from monitoring compliance with command and control regulations toward site-specific monitoring of compliance according to agreed service provision measures or monitoring of actual service provision. This facilitates a learning process as suggested by adaptive IWRM approaches.

1 Resource costs are defined by the European Commission (EC) as "the costs of foregone opportunities which other users suffer due to the depletion of a resource beyond its natural state of recharge or recovery" (e.g. groundwater mining) (WATECO, 2003). In 2004, the definition of resource costs was extended by the European working group ECO2 by adding the costs of inefficient allocation of water by existing institutions (Brouwer, 2004).

2 Environmental costs are defined by the EC as "the costs of damage that water uses impose on the environment and ecosystems and those who use the environment" (WATECO, 2003). According to recommendations of the ECO2 working group, these costs should be measured in terms of lost environmental benefits, where these benefits are measured using environmental valuation methods such as contingent valuation, travel cost methods, or hedonic pricing (Brouwer, 2004).

The PHES implementation practices reveal that intermediaries play an important role for the design and execution of schemes. These intermediaries can be divided into two principal types. During the preparatory design phase of PHES schemes, intermediaries often act as facilitators or knowledge providers in order to reveal bio-physical cause-effect relationships which functionally connect service providers, i.e. land users, with beneficiaries and to create an enabling environment. In the subsequent execution phase, intermediaries are usually involved as moderators, trusted managers of funds and technical supervisors or advisers. The intermediary type which is present during the preparatory phase is often an external organization, e.g. an international donor agency or an agency of the national government, while the intermediary during the execution phase is generally made up of local stakeholders and trusted organizations. Hence, the former is of a transitory kind and the latter is enduring, possibly, by establishing an intermediary structure to form the basis for RBOs or RBCs.

What are the contributions of PHES schemes in a specific empirical context of IWRM implementation?

The Gíl González PHES scheme implementation in Nicaragua could demonstrate that horizontal interplay across sectors and municipal boundaries can be achieved from bottom-up, although top-down structures, e.g. an RBO or RBC, are still missing. The Gíl González PHES scheme successfully initiated IRBM at the sub-basin level through active stakeholder involvement, basin-wide land use planning and the establishment of an RBC. Moreover, municipalities were strengthened in their role as local environmental managers and attained co-financing from a private water user. Additionally, the Gíl González PHES scheme could attract additional funding from the FONADEFO and the FAO-Facility network. This co-financing is a significant contribution to achieve effective decentralization of the IWRM process through territorial planning, catastral development and land use planning at the sub-basin level.

Besides improving the IWRM operationalization from bottom-up, the PHES instrument plays an important role in relation to other environmental regulation. The case study from Nicaragua could demonstrate that the implementation of the PHES scheme has led to diffusion and increased acceptance of existing formal rules on land uses and enabled a redefinition of traditional roles in environmental policy-making. The assessment of the Gíl González PHES scheme, in this dissertation, could demonstrate a significant contribution to the co-management of water resources at the operational level. Through the bio-physical assessment and the identification of land as well as water uses fundamental conditions to initiate actual natural resource management could be achieved. Hence, local actors assumed tasks which according to the water law were assigned to the ANA. Because of a variety of constraints, the ANA is still not able to fulfill these tasks.

In order to take advantage of local PHES initiatives, the ANA could actively promote PHES development from bottom-up. Hence, the ANA could take the role of an intermediary to ensure that bottom-up development meets top-down requirements. Moreover, recommendations for the establishment of river basin management plans could assign local initiatives to assist ANA with its tasks, e.g. bio-physical assessment, registration of land and water use, economic valuation, and the creation of organizational structures in form of an RBC. These opportunities to link local and national policies may be taken up in the context of the Law for the National Water Fund, the Law on PHES and the National Registry of Water Users which are still to be established. Furthermore, the knowledge gained through local PHES schemes could also be incorporated in the National Environmental Information System (Sistema Nacional de Información Ambiental; SINIA).

The inclusion of PHES schemes as a management instrument in the National Water Law in Nicaragua is probably an advantage for the further dissemination of such schemes, although a specific law on PHES still needs to be passed. For the design of this specific law, it seems reasonable to build on experiences and lessons learned from PHES schemes already implemented. The typical implementation steps and different roles of actors, as identified and analyzed in this dissertation, deserve special attention. The PHES law and the law which will establish the National Water Fund provide opportunities to integrate PHES schemes more explicitly in the operationalization of IWRM, thus, describing their functional role in this process.

Can general recommendations be made on how PHES may contribute to the IWRM process?

Nicaragua represents a typical example for IWRM implementation in developing countries characterized by improvements in horizontal interplay at the national level, formal provisions for vertical interplay and improvements of fit through RBOs, but incomplete implementation because of resource constraints. Other developing countries can probably learn from Nicaraguan PHES experience. The facilitation of local PHES development as part of IRBM at the operational level can represent an attractive option to local actors. Facilitation could formally be achieved, for instance, by including PHES, or at least the assessment of hydrological ecosystem services and their monitoring, in the development process of river basin management plans. However, mechanisms for upward connectivity, i.e. vertical interplay from bottom-up, are needed to link local IWRM progress at the lower operational level with IWRM at higher operational and strategic levels. This linkage can usually be provided through membership linkage, thus, strategic stakeholders within a PHES scheme, e.g. municipalities, automatically become members of higher level organizations. These higher level organizations, RBOs for instance, can serve as a diffusion platform of best practices and may coordinate different local activities.

However, it is important to stress that the PHES instrument is only one option among other legislative measures or infrastructures to substitute hydrological services more efficiently. PHES schemes can have negative environmental and social (e.g. on equity issues) impacts and combinations with other efforts will often be required to avoid them. As a policy mix, in combination with command and control instruments, PHES schemes can be applied to conservation actions to achieve a state above a commonly agreed *duty of care*. Command and control instruments would then be applied below compliance of this *duty of care*. Especially in developing countries, where the rules in use often differ significantly from formal rules, it is important that a rule-making process takes place to define the *duty of care* standard, as shown by the implementation of local PHES schemes. This standard may develop dynamically providing a smooth transition towards actual compliance with formal command and control regulations. Intermediaries can play an important role as moderators to ensure equal participation of weaker actors and as knowledge providers with regard to existing formal rules.

The establishment of (local) PHES schemes requires time, leadership, particular biophysical and social conditions, and has to fit with national and regional laws. The initial costs for bio-physical assessments, economic valuation, stakeholder involvement and persuading efforts to gain sufficient support for the scheme development involve large upfront costs. Favorable bio-physical and socio-economic river basin features are given when downstream service users are able to pay and actually depend upon hydrological ecosystem services for their economic, social or cultural well-being. In most cases this implies a particular awareness of the existing dependence or a perceived responsibility for environmental stewardship, i.e. corporate responsibility in the case of enterprises. Moreover, services are often already in a state of deterioration or are threatened to cease altogether. Such perceived threats can drive the incentive to invest in conservation measures.

Often service beneficiaries need to deem compensations for service provision as being justifiable in order to be willing to pay, thus, social relationships between upstream and downstream actors can play a significant role for the willingness to join a scheme. In many cases, downstream beneficiaries feel solidarity with poorer upstream land users. Potential service buyers are generally enterprises, that use water as a significant production factor, or public authorities, as is often the case in water supply. Indeed water suppliers represent the largest group among all service buyers. But as different PHES schemes in Nicaragua have demonstrated, public water suppliers are often difficult to engage. Moreover, since the water supply is often highly subsidized in developing countries, payments for hydrological ecosystem services need to be justified against other costs that need to be covered as well, e.g. investments in infrastructure and maintenance or the real cost of water production. Many water users in developing countries cannot pay the actual cost of water supply. Here the situation is very different from the one in developed countries where those costs are often better covered by existing water prices. Nevertheless, PHES can serve as an important communication tool to stress the value of ecosystems.

As resources for IWRM operationalization are scarce, considerations beyond the hydrological ecosystem service concept in terms of cost and benefit sharing through PHES schemes are justified. Nevertheless, payments do not have to be monetary and may instead take the form of material compensations or even social appreciation. In the Andes, reciprocal agreements, for instance Reciprocal Water Agreements in Peru (Martinez et al., 2013), have developed because there was no social acceptance for monetary payments. It has even been argued that monetary payments may potentially erode other motivations for conservation. At best, mutual agreements on the specific form and extent of compensations are reached between providers and beneficiaries. Often this increases the sense of ownership among participants.

The degree of stakeholder interaction and involvement in decision-making certainly depends on the geographical size of action. For small-scale local PHES stronger stakeholder involvement and participation in scheme design and implementation is possible compared with larger scales. In purely national schemes, the other extreme to local ones, there is often no participation at all in the scheme design and very limited participation, often only on the provider side, in the implementation process. However, the experience with water funds in South America raises hopes of striking a balance between local participation and a larger spatial impact. This often implies that provider and beneficiary groups take some form of social organization through group representatives. At best, these forms of social organization already exist and have been legitimized. An example of such an organization is the association of plantain producers in the Gíl González river basin.

In Mexico, experiments were made with a combination of local and national funding for PHES schemes in the form of water funds (FIDEICOMISO). If payments from local beneficiaries are insufficient to engage sufficient service providers, this combined financing can provide a solution. Moreover, financial contributions from other policy levels are justified since benefits are derived not only locally and regionally but also nationally and beyond. A mix of funding sources may be facilitated by considering additional ecosystem services such as biodiversity conservation (IPBES, TEEB) and carbon sequestration (REDD) as a bundle. However, for the operational IWRM hydrological ecosystem services are most appropriate.

Complex bio-physical cause-effect relationships which are often not well understood and difficult to communicate can be a further limitation to the application of PHES schemes. In many societies, forests are considered as especially valuable for the provision of hydrological ecosystem services, however, other ecosystems may often also provide these services. Although PHES schemes in practice do not follow a strict market logic and rather focus on broader stakeholder involvement and participation, efficiency and equity issues need to be carefully balanced for the long-term success of schemes. This means that the areas with the highest service provision potential and the lowest relative opportunity costs should be given priority. In the end, to be successful the often assumed cause-effect relationships need to be verified. Hence, for accountability purposes the monitoring and measuring of service provision is necessary. Hydrological models may support the bio-physical assessment and monitoring if sufficient information is available to run them. However, not all hydrological models can be used to provide information about hydrological ecosystem service at any desirable site so that provision and benefits can be identified.

While water resources are the integrating factor within IWRM, the ecosystem service concept establishes hydrological ecosystem services as the integrator of different sectors and administrative units. This places stronger emphasis on the interactions between land and water users. While the concept of hydrological ecosystem services considers only the supply side, it is also conceivable to place stronger consideration on the demand side by looking at competing water uses representing different service beneficiaries. Leaving water in the river can also be considered as a hydrological service and worthy of compensation, either by downstream beneficiaries of this service or others. Experiences with PHES schemes aiming at reducing water uses to the benefit of others have been made in the USA and Australia. Thus, in principle, PHES schemes may also work among competing beneficiaries.

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Vita

Ein Lebenslauf ist in der elektronischen Version dieser Dissertation aus Gründen des Datenschutzes nicht enthalten.